ECOLOGICAL COMMUNITY CHANGES AND ECOSYSTEM FUNCTIONING DURING STRUCTURED WETLAND DEVELOPMENT

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ABSTRACT

Saltmarshes provide important ecosystem services, including protection from tidal inundation, grazing for livestock, resources for birds and breeding grounds for fish. After a long history of saltmarsh reclamation there is now emphasis on restoration of existing or destroyed saltmarshes to provide dynamic coastal defences providing ecosystem services. The engineering technique of Managed Realignment (MR) is a popular method of restoration where existing coastal defences are deliberately breached. The process of change associated with MR, including the influences of starting state, construction, and saltmarsh community structure changes require further study to provide guidelines for optimal restoration practices. Here, the ecological changes during the early stages of saltmarsh restoration via MR at Steart Marshes in Somerset, UK are studied. Common Standards Methods were used to monitor vegetation and invertebrate community composition changes from pre-breaching (September 2014) through 4 years of tidal regime post-breaching. Soil and sediment characteristics, and soil accretion were studied at the same time. All data were collected from permanent sample plots. Changes were compared with a managed brackish site (Otterhampton Marsh) with a similar starting state, and a mature SSSI saltmarsh in Bridgwater Bay was treated as a 'target state'. Soil characteristics included soil moisture, pH, electrical conductivity, soil organic carbon, nitrate-nitrogen, ammonium-nitrogen and phosphate-phosphorus. Changes of these were compared to the target state and the managed brackish site. The National Vegetation Classification (NVC) scheme was partly used to assess vegetation communities, and multivariate methods including cluster analysis and ordination were used to assess changes of flora, invertebrates, and soil characteristics. Saltmarsh vegetation colonised a former pasture location more rapidly than a former arable location and was slowest in the managed brackish location. Colonisation by invertebrates and soil characteristics followed a similar pattern. The most abundant invertebrate species were identified to species and quantified, and their abundance on Steart Marsh was compared to the mature SSSI saltmarsh and brackish site. The three most abundant species overall were *Orchestia gammarellus*, *Campiglossa plantaginis*, and the invasive planthopper, *Prokelisia marginata*. *C. plantaginis* was positively correlated with the abundance of colonising *Aster tripolium,* and *P. marginata* had a positive relationship with colonising *Spartina anglica.* It is likely that soil compaction, partly caused by pre-breach engineering, influenced differences in soil characteristics between study sites and in plant and invertebrate colonisation, principally due to the prevalence of standing water associated with anaerobic conditions. Compaction and land height (AOD) influenced plant colonisation across all sites and soil moisture and electrical conductivity influenced colonisation by plants and invertebrates on Steart Marsh and Otterhampton Marsh after the breach. Soil carbon storage was greatest in the former pasture site. Vegetation, invertebrates and soil characterstics in the former pasture site on Steart Marsh started to resemble that of the SSSI saltmarsh three years after tidal inundation commenced, and overall the results indicate that restoration is influenced by former site use and engineering practices involved in Managed Restoration.

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CHAPTER 1: INTRODUCTION

Wetlands are defined as ecosystems that periodically flood (Brinson, Lugo and Browrz, 1981) and include coastal marshlands and inland wetlands, such as lakes, rivers and swamps. They can provide ecosystem services, including economic value to local populations (Mitsch and Gosselink, 2000; Schuyt, 2005), and importantly, direct mitigation of climate change via carbon sequestration (Burden *et al.*, 2013; Mitsch *et al.*, 2013; Beaumont *et al.*, 2014). Wetlands have a considerable effect on the hydrological cycle (Bullock and Acreman, 2003), providing flood prevention if managed correctly (Bullock and Acreman, 2003; Zedler and Kercher, 2005). However, it is estimated that since 1900 64-71% of the world's wetlands have been lost due to human activity (Davidson, 2014); chiefly due to land reclamation, over-exploitation of the resources they offer, pollution, and anthropogenic changes in hydrological processes (Junk *et al.*, 2013).

Historically, wetland sites have been used by humans for agriculture, habitation and recreation, which has led to the modification and degradation of such sites and the introduction of toxic materials and invasive species (Michener *et al.*, 1997). However, the importance of wetlands for providing ecosystem services is now widely recognised, which means there is an increasing effort towards restoring both coastal and inland wetlands so that they can once again provide the ecological functions that they performed before the impact of significant detrimental human encroachment.

Coastal wetlands include saltmarshes, mudflats, swamps, seagrass beds, freshwater marshes, and mangroves. Worldwide, the impact of human activity has depleted over 90% of important species in coastal and estuarine habitat, as well as causing a reduction of water quality, and hastening species invasions (Lotze *et al.*, 2006). Recent research has concentrated on the ability of coastal wetlands to store carbon and prevent flooding of local communities. The carbon that these ecosystems can sequester has been termed blue carbon and enhancing or maintaining the carbon storage capacity of coastal wetland soils has a role in reducing atmospheric carbon and meeting the targets of the 2015 Paris Agreement.

1.1 Defining saltmarshes

Saltmarshes are dynamic systems that form between the sea and the land. They provide essential habitat for plants and animals, many of which are valuable food sources (Chmura, 2013), and they act as a buffer to terrestrial habitats, including reducing flood risk (Foster, Hudson and Bray, 2013). Furthermore, they provide a number of other benefits to humans, such as controlling erosion, purifying water, sequestering carbon, supporting

fisheries, and providing recreation, education and research possibilities (Gedan, Silliman and Bertness, 2009; Barbier *et al.*, 2011; Townend *et al.*, 2011; Chmura, 2013; Beaumont *et al.*, 2014; Rogers *et al.*, 2016). Historically, the degradation of coastal wetlands has been at a slower rate than inland wetlands, but the loss of the former still remains high (Davidson, 2014). It has been suggested that saltmarshes are being degraded or lost to a phenomenon known as coastal squeeze (Doody, 2004; Hughes, 2004), where rigid barriers, such as sea walls and embankments, block the landward migration of saltmarsh species (Ross and Adam, 2013; Martínez *et al.*, 2014). As sea levels are rising due to climate change, saltmarshes are likely to shrink or disappear (Torio and Chmura, 2013), but there is recent evidence to suggest that they may be resilient to this change (Schuerch *et al.,* 2018). Nevertheless, relative mean sea level is rising (Robins *et al.*, 2016), and it is apparent that this will diminish most saltmarshes' capability of maintaining adequate elevation through accretion (Gedan, Silliman and Bertness, 2009; Crosby *et al.*, 2016), especially in Southern England and Wales where isostatic land recovery has not exceeded eustatic sea level rise during the Holocene (de la Vega-Leinert and Nicholls, 2008). It is also expected that wave energy generated by rising wind strengths will continue to contribute to increased erosion of saltmarshes throughout the current century (Jones *et al*., 2013).

The increasing cost of maintaining sea walls is becoming substantially more challenging, so ecosystem creation or restoration is an appealing strategy (Temmerman *et al.*, 2013). Moreover, the building of higher sea walls may in fact facilitate an increase in coastal squeeze (Boorman, 2003), which will exacerbate the problem. It is expected that restoring tides to reclaimed saltmarshes will return ecological functions and attributes to these areas (Warren *et al.*, 2002), although the time it will take to achieve this goal is debated in the literature.

1.2 Saltmarsh restoration

There are various methods of saltmarsh restoration, such as transplanting vegetation or installing wave breaks, but a soft engineering technique called managed realignment (hereafter MR) has become popular (Paramor and Hughes, 2005). This method involves deliberate breaching or removal of existing coastal defences with the purpose of re-creating intertidal habitat (Paramor and Hughes, 2005; Esteves, 2014; Figure 1.1).

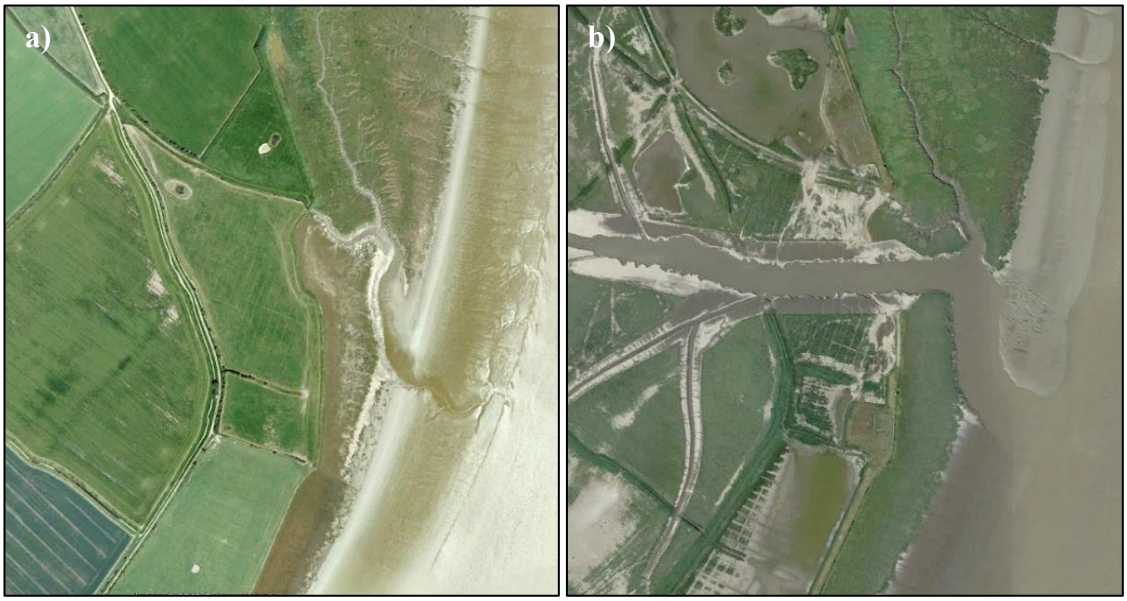


Figure 1.1 Managed Realignment at Steart Marsh, Somerset: a) MR site prior to creation, showing an embankment between agricultural land and an estuary; b) MR site after breaching the embankment, with estuarine water entering a constructed creek system (Google Earth, 2018).

De-embankment through MR allows coastal habitats to migrate landwards in response to sea-level rise (Pontee, 2014), and in recent years such schemes have been used across Europe and the United Kingdom to mitigate the loss of mudflat and saltmarsh ecosystems (Morris, 2016). The primary aim is to reduce the length of required sea defence and increase the area of intertidal habitat so that it can once again provide important ecosystem services, especially flood defence (Andrews *et al.*, 2006). There are a number of methods of implementing MR, including the removal, breach or realignment of defences, controlled tidal restoration (tidal flow controlled through culverts and sluices) and managed retreat, which involves planned retreat from areas prone to flooding (Esteves, 2014). In the last few decades there have been over 50 MR schemes in the UK using a variety of these implementation methods (Pontee, 2014). The creation of new habitats through these schemes is also in response to the European Union Habitats and Birds Directives (Atkinson *et al.*, 2004). It is a requirement under these directives that Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) are designated across European Union territories to decrease damage to European natural habitats and their related species (Airoldi and Beck, 2016), such as shore birds and waders.

It is suggested by Esteves (2013) that the value of MR as a sustainable coastal management option is associated with the multiple functions it can provide, but it is important to determine and assess changes in the biotic and abiotic structure of restored saltmarsh ecosystems, because this will lead to an understanding of the associated functions

that are essential in the provision of ecosystem services. Importantly, saltmarsh ecosystem dynamics can be affected by historic land uses (see section 1.3.3.1), which has been evidenced by Almeida *et al.* (2014) in a number of saltmarsh habitats in Portugal. There is evidence to suggest that coastal plant species do colonise MR sites rapidly, but community composition may not be the same as reference saltmarshes, and pioneer species may remain dominant (Mossman, Davy and Grant, 2012).

1.3 Re-establishing ecological processes on restored saltmarshes

1.3.1 Saltmarsh flora

Self-sustaining plant communities are an important goal of saltmarsh restoration because plant communities have both biological and economically important functions and are a good indication of the performance of restored marshes (Garbutt and Wolters, 2008). Elevation is a key factor that influences the establishment of salt-marsh vegetation (Wolters, *et al.*, 2008) and saltmarsh plants tend to show a vertical zonation (Figure 1.2) where the lower limits of the different species are determined by tolerances to physical factors associated with immersion, such as low pH and anoxia of the soil (Hughes and Paramor, 2004), and upper limits are influenced by competition.

Figure 1.2 has been removed from this version of the thesis due to copyright restrictions

Figure 1.2 Typical saltmarsh zonation (Chirol et al., 2018; modified from Foster et al., 2013).

The local diversity of saltmarsh plant communities is relatively low (Silliman, 2014) because the essentially terrestrial vegetation has to be tolerant of salinity and submergence from the tide. The plants that succeed under these stressful conditions are known as halophytes. Some species are obligate halophytes that require a certain level of salinity to survive, whereas others are non-obligate halophytic species that are tolerant of saline conditions but can survive quite readily without them.

According to Rodwell (2000), there are 28 saltmarsh plant communities in Great Britain, and the most common herbaceous species found on the seaward edges of European temperate-zone saltmarshes are the grasses, *Puccinellia maritima* and *Spartina anglica* (Gray and Mogg, 2001; Figure 1.3). Other common plants of the lower marsh are, sea aster (*Aster tripolium)*, glasswort (*Salicornia spp*.), scurvy grass (*Cochlearia spp.)*, annual sea blite (*Suaeda maritima)*, and sea spurrey (*Spergularia spp*.). Spear-leaved orache (*Atriplex prostrata*), and sea purslane (*Atriplex portulacoides*) are common in the middle marsh, and sea couch (*Elytrigia atherica)* usually occurs in the upper marsh, especially above mean high water spring tide level (MHWS).

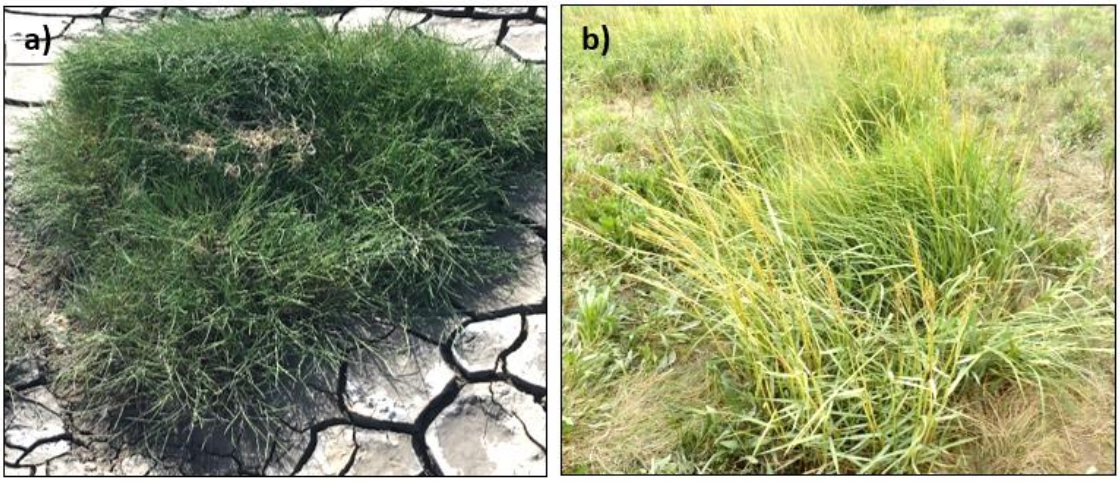


Figure 1.3 a) *P. maritima* at Steart Marsh, Somerset; b) stand of *S. anglica* at Steart Marsh, Somerset. Photograph taken by A. George (2016).

Colonisation processes may be influenced by species characteristics. At Abbotts Hall and Tollesbury MR sites in Essex (UK) it was discovered that *Salicornia europaea* and *S. maritima* were early colonisers, but they were slowly replaced at higher elevations, by *S. anglica*, *P. maritima*, *A. portulacoides* and *E. atherica*, which colonised from low to high saltmarsh (Hughes, Fletcher and Hardy, 2009). In a study in the Netherlands, it was found that the seeds of *A. tripolium*, *A. portulacoides*, *P. maritima* and *S. anglica* may not have a good capacity to float, whereas the seeds of *Spergularia maritima*, *Salicornia* spp. and *S. maritima* can float well over small distances (Wolters and Bakker, 2002). This may suggest why *S. europaea* and *S. maritima* were the early colonisers at Abbotts Hall and Tollesbury.

It has been observed that both common saltmarsh-grass (*P. maritima*) and common cord-grass (*S. anglica*) prevent the re-suspension of sediment which facilitates accretion (Gray and Mogg, 2001). For example, Langlois, Bonis and Bouzillé (2003) found that soil covered with *P. maritima* almost doubled in height annually compared to areas with bare

sediment. *P. maritima* is a stoloniferous C_3 (efficient at photosynthesis in temperate climates) perennial species, whereas *S. anglica* is one of the few C_4 rhizomatous perennial species that resides naturally in temperate zones (Dunn *et al.*, 1987; Huckle, Potter and Marrs, 2000).

The *S. anglica* dominated National Vegetation Classification (NVC) community is SM6, but the species is found in almost all saltmarsh communities (Adam, 1981). It originated in England during the late 1800s, following chromosome doubling in *Spartina x townsendii*, a hybrid between the British native *Spartina maritima* and *Spartina alterniflora*, which was introduced from North America (Raybould *et al.*, 1991). According to Adam (1990), *S. anglica* can tolerate tidal submergence more than any other saltmarsh plant species found in Europe, thus it can outcompete other species at the same elevation, such as *Salicornia*. However, pairwise experiments with *Spartina* and *Puccinellia* under different environmental conditions (salinity, sediment and waterlogging) have shown that *Puccinellia* exerts above ground competitive dominance over *Spartina* in relation to all sediment types, apart from in sand, and when immersion levels are low (Huckle, Potter and Marrs, 2000). *S. anglica* can be beneficial because it reduces erosion, increases productivity and creates grazing marsh, but it can become a monospecific sward, which has an impact on saltmarsh plant diversity and bird feeding grounds (Doody, 1990). Although the number of saltmarsh species is relatively low, plant diversity is a key aspect of saltmarsh systems because it can increase biomass, create complex canopies and facilitate seedling recruitment (Zedler, Callaway and Sullivan, 2001). Therefore, it is important to create conditions on an MR site that facilitates plant colonisation and diversity.

The establishment of seeds on MR sites is a crucial factor in plant community restoration. It has been suggested that a mature saltmarsh next to an MR site is essential for rapid colonisation by saltmarsh plants (Erfanzadeh *et al.*, 2010), and factors such as accretion and erosion can have an influence on the establishment of seedlings, especially if these changes are abrupt rather than gradual (Cao *et al.*, 2018), which they are likely to be on an MR site. Wolters, Garbutt and Bakker (2005) recommend that the optimum time to breach an MR embankment is before or during September due to the peak in dispersal of saltmarsh species, although issues can arise with colonisation of plants on MR sites. For example, Sullivan *et al.* (2018) suggest that plant species, such as *A. portulacoides,* may establish outside of their normal niche on restored sites due to the prevalence of bare ground after breaching, which can impede the establishment of characteristic species due to competition. On an MR site in Brancaster, Norfolk, UK it was found that plants colonised rapidly, but

plant communities were still dissimilar to reference marshes after 5 years, and bare ground remained prevalent (Mossman *et al.*, 2012). If plant communities on MR sites are dissimilar to reference marshes, it is unlikely that they will be functionally equivalent (Mossman, Davy and Grant, 2012). Therefore, accretion, seed availability, and the development of suitable soil characteristics on MR sites need to be considered if the aim is to establish a functioning marsh with a greater diversity of vegetation, which will also increase colonisation potential and habitat suitability for a range of fauna.

1.3.2 Saltmarsh fauna

1.3.2.1 Invertebrates

Invertebrate macro-fauna is an important component of the saltmarsh system, primarily because a number of species aerate saltmarsh soil through burrowing and they incorporate surface organic matter into the soil (Boorman, 2003). Herbivorous invertebrates also have strong control over marsh plant productivity (Silliman and Bortolus, 2003), and they are important food sources for vertebrate species. As saltmarshes are dynamic habitats with environmental constraints, there may be a low number of invertebrate species present, but the populations of the species usually consist of a large number of individuals (Teixeira, Duarte and Caçador, 2014). Marine derived invertebrates, such as crustacea and molluscs, can especially tolerate tidal immersion, and their populations on marshes are often high (Nottage and Robertson, 2005). Common ground-dwelling saltmarsh invertebrate species in the United Kingdom include those within the carabid family (ground beetles), Araneae species (true spiders), isopods, and amphipods such as *Orchestia gammarellus* (Ford *et al.*, 2013).

Orchestia gammarellus (Figure 1.4) is a semi-terrestrial amphipod in the family Talitridae, and it is commonly known as a sandhopper or beachflea. It is an important species to the saltmarsh ecosystem as it is a food source for commercially important fish species such as sea bass (Cattrijsse and Hampel, 2006), and it plays a vital role in nutrient cycling and aeration of the soil (Schrama *et al.*, 2015). The species has been described as the earth worm of the saltmarsh because, as a macrodetrivore, it fills a similar ecological niche (Schrama, 2012), and a field experiment by Schrama *et al.* (2015) has shown that the species can stimulate nitrogen mineralisation and accelerate vegetation succession.

Figure 1.4 Amphipod species *O. gammarellus*, commonly found in saltmarsh habitats in North-West Europe. Photograph taken by A. George (2016).

Previous research has shown that *O. gammarellus* becomes the dominant invertebrate species on mature saltmarshes of the detritus-based food web (e.g. Schrama, Berg and Olff, 2012), so it is reasonable to suggest that *O. gammarellus*, and perhaps other intertidal amphipod species, are important for the functionality of marshes, and the presence of such species should be assessed when considering saltmarsh restoration or creation.

According to Adam (1990) approximately 130 of the insect species found on saltmarshes in the Wadden Sea are phytophagous, with groups such as Cicadellidae (leafhoppers), Aphidoidea (aphids), Coleoptera (beetles), Lepidoptera (particularly moths), Cecidomyiidae (gnats), and other species of flies feeding on halophytes including *P. maritima* on the lower marsh. Plant hoppers (Delphacidae) may also be prevalent on marshes, and the invasive *Prokelisia marginata* (a native of North America) is being found in increasing numbers on *Spartina* marshes in the UK (Harkin, 2016). Many different fly species accumulate on saltmarshes, particularly in the spring and summer, including Syrphidae (hoverflies), Dolichopodids (long-legged flies) and Tipuloidea (craneflies) (Boorman, 2003). The phytophagous dipteran species, *Campiglossa plantaginis* (formerly *Paroxyna*), is also common on saltmarshes and it has an association with sea aster (Albrectsen, Ericson, and Lundberg, 2008). There are also a number of ground beetles that utilise saltmarsh habitats along with specialist rove beetles (Boorman, 2003). Within saltmarshes, ground beetles tend to be more restricted to specific vegetation sub-communities than spider species, but both taxa may be scarce in the lower marsh (Irmler *et al.*, 2002).

There is a relative lack of information related to saltmarsh spider communities, especially in comparison to the extensive studies based on communities in agricultural systems. Nevertheless, it is clear that many species have been found in saltmarsh habitats (e.g. Ford *et al.,* 2013), and there is evidence from Essex, UK where 43 spider species were

found on a combination of MR sites, accidently realigned sites and saltmarsh reference sites (e.g. Petillon *et al.*, 2014). Some of the spider species found on saltmarshes are coastal specialists, such as the wolf-spiders (Lycosidae) *Pardosa purbeckensis* and *Arctosa fulvolineata*. *P. purbeckensis* is more widespread than *A. fulvolineata* in coastal habitats of the United Kingdom, and it is often confused with *Pardosa agrestis,* which is a terrestrial species. However, the form *P. purbeckensis* is almost exclusively found in habitats that are high in salinity (Pétillon *et al.*, 2011). Although the species is a ground-dwelling invertebrate, previous research has suggested that it avoids flooding by climbing up vegetation during high tides (Pétillon, Montaigne and Renault, 2009). *P. purbeckensis* spiderlings hatch from June to August (Bonte, Bossuyt and Lens, 2007), and they predate on other invertebrates on the marsh. The species is an important food source for birds, such as meadow pipets that feed on saltmarsh invertebrates (van Klink *et al.*, 2013). Ford *et al.* (2017) discovered that ground running hunters, such as *P. purbeckensis*, were positively associated with plant cover, height, and canopy complexity during summer, which further supports the need for the development of heterogenous plant communities on MR saltmarshes. However, research by Petillon and Garbutt (2008) showed that the species was more abundant on MR sites in the Blackwater River Estuary in Essex than on natural reference marshes, possibly due to the open canopy on MR sites comprising a mixture of bare ground and plants, rather than the closed canopy found on mature marshes.

Money spiders (Linyphiidae) are also common in saltmarsh habitats. These include the black and orange money spider, *Oedothorax fuscus*, which can also be found in other habitats, and the coastal specialist sheet weaver, *Erigone longipalpis* (Ford *et al.*, 2013). Although there are numerous spider species that may be captured on saltmarshes, Ranwell and Ratcliffe (2012) note that *P. purbeckensis* and *E. longipalpis* are the most widespread and characteristic species. Spider distribution, abundance, and community composition may not just be affected by vegetation structure, but also by elevation of the marsh (Döbel, Denno and Coddington, 1990), so these factors should be taken into account on MR sites if it is desirable to achieve similar species assemblages to those found on mature saltmarshes.

In 2003 the Paull Holme Strays MR site in the Humber estuary, UK was breached, and researchers discovered that there was low abundance and diversity of macrofaunal communities initially, but benthic invertebrate community biomass increased substantially between 2004 and 2005, although it was noted that colonisation was concentrated predominately around the breach sites (Mazik *et al.*, 2007). According to Atkinson *et al.* (2004) the first benthic invertebrate species found in large numbers at the Tollesbury (Essex)

MR scheme after the breach in the 1990s were the mudsnail (*Hydrobia ulvae*), Baltic clam (*Macoma balthica*), paddleworm (*Eteone longa*), catworm (*Nepthys hombergi*), the polychaetes *Nereis diversicolor* and *Pygospio elegans*, and the bristleworm species, *Spio filicornis*. In the Orpland MR site in Essex, fewer species colonised, but *H. ulvae* and *N. diversicolor* were common and widespread (Atkinson *et al.*, 2004). At Tollesbury it was found that invertebrates only colonised the newly accreted sediments of the MR site and were not present in the original agricultural soils (Garbutt *et al.*, 2006), and Evans *et al.* (1998) found that the colonisation of benthic invertebrates, such as *Hydrobia* and *Nereis*, on a restoration site in North-East England was slower than anticipated due to soil compaction produced by earth moving equipment used during the construction phase of the restoration scheme and low organic matter in the substrate. However, an increasing body of research suggests that *Hydrobia* and *Nereis* may have a negative impact on saltmarshes (e.g. Andersen, 2001; Widdows and Brinsley, 2002; Paramor and Hughes, 2004; Widdows, Brinsley and Pope, 2009). *N. diversicolor* in particular may impede plant succession through feeding activity (Paramor and Hughes, 2004), which may have implications for the development of vegetation on MR sites. The species can also cause erosion of saltmarsh sediments due to feeding and burrowing (Paramor and Hughes, 2004; Widdows, Brinsley and Pope, 2009), and it has been questioned whether marsh losses could be due to the effects of this rather than sea level rise and coastal squeeze (Paramor and Hughes, 2004).

H. ulvae is usually widely distributed over saltmarshes, and it feeds as a surface-deposit feeder and an epipsammic algal grazer on particles that cannot be ingested (Frid and James, 1988). Although the species is an important part of the saltmarsh food web, it has been reported that large populations of this species can also increase the erosion rate of saltmarsh sediment (Andersen, 2001; Widdows and Brinsley, 2002). Additionally, there is evidence that macro-invertebrates can be responsible for die-back of vegetation (Silliman and Bertness, 2002; Silliman *et al.*, 2005; Holdredge, Bertness and Altieri, 2008), although it has been argued that these effects could be local and not solely responsible for sudden die-back (Alber *et al.*, 2008). Despite possible negative implications related to the physical structure of saltmarshes, invertebrate species such as *Nereis* and *Hydrobia* may contribute to a disturbance regime that maintains the temporal dynamic of saltmarsh communities, and they also constitute part of the diets of wildfowl and shorebirds. Therefore their presence on MR sites may be necessary to attract sediment probing bird species such as grey plover (*Pluvialis squatarola*), shelduck (*Tadorna tadorna*), dunlin (*Calidris alpine*) and common redshank (*Tringa tetanus*) (Atkinson, 2003).

1.3.2.2 Birds

Saltmarshes are used by birds for feeding (on flora or fauna depending on the species of bird) and nesting, and these ecosystems are especially important for wading species such as common redshank that use them for breeding grounds (Hughes, 2004); however, it is important to note that waders feed mainly on mudflats rather than saltmarshes specifically (Adam, 1990). Fuller (1982) suggests that common birds found on British saltmarshes include colonial and non-colonial species, middle and upper marsh species, species which occupy grazed upper saltmarsh, fringe species that occupy the transitional zone (Table 1.1), and species such as shelduck that use marshes for nesting. Oystercatchers (*Haematopus ostralegus*) are also found on saltmarshes, and wintering birds such as geese commonly feed on *P. maritima* (Allen and Pye, 1992).

Table 1.1 Common birds found on British saltmarshes (modified from Fuller, 1982).

Saltmarsh bird category	Examples
Colonial	Gulls and terns
Non-colonial	Waders (e.g. redshank)
Middle and upper marsh species	Meadow pipit (*Anthus pratensis*) Reed bunting (*Emberiza schoeniclus*) Skylark (*Alauda arvensis*)
Grazed upper saltmarsh	Lapwing (*Vanellus vanellus*) Dunlin (*Calidris alpina*)
Fringe species	Moorhen (*Gallinula chloropus*) Sedge warblers (*Acrocephalus schoenobaenus*)

Sea level rise will have an influence on coastal bird populations and communities if there is a decrease of this important intertidal habitat (Norris, Atkinson and Gill, 2004). Therefore, the creation or restoration of saltmarsh can provide habitat for many obligate avian species as well as facultative species, which utilise other habitats as well (Lewis and Casagrande, 1994). However, saltmarshes are subject to changing tidal levels and salinities, and if these factors are not comprehensively addressed during restoration there may be a lack of diversity on the marsh (Atkinson, 2003). Past land-use and livestock grazing on restored sites may also have an influence on the ecological processes that occur (Spencer and Harvey, 2012; Clausen, Stjernholm and Clausen, 2013), which could have an influence on the colonisation of both plants and animals, including wetland bird species.

1.3.2.3 Saltmarsh grazing

It is apparent that grazing can have both negative and beneficial effects on saltmarsh habitats and associated species. It has been evidenced that heavy grazing by terrestrial vertebrates can be a threat to breeding redshank (Norris *et al.*, 1998) and can have a damaging influence on the feeding activity and growth rate of fish such as sea bass (Laffaille, Lefeuvre and Feunteun, 2000). However, livestock grazing can be beneficial to geese species (Bakker, Bos and Vries, 2003; Bos *et al.*, 2005), and many inter-tidal coastal specialist invertebrate species have been shown to prefer grazed marsh (Ford, Garbutt, Uren, *et al.*, 2013). However, *P. purbeckensis* numbers decline in grazed saltmarshes due to the lack of vegetation heterogeneity (Pétillon *et al.*, 2007). The benefits are therefore debatable, with some researchers advising that there should be cessation of grazing on saltmarsh habitat (see Kiehl *et al.*, 1996) and others suggesting that a rotational grazing policy should be explored so that certain species of arthropod are not lost (see van Klink *et al.*, 2013). It has also been suggested that low stocking densities of livestock such as sheep should be considered so that the invasive grass *Elymus athericus* does not increase, and typical halophilic plant species are not lost (Pétillon *et al.*, 2007). Grazing by livestock on restored or created marshes can be a useful management technique, due to their effect on soil and species composition, but stocking density, type of stock, and time of release onto the marsh need to be considered to ensure that negative impacts, such as poaching and overgrazing, are reduced (Clausen, Stjernholm and Clausen, 2013).

1.3.3 Saltmarsh soil characteristics

Primary production is usually high in saltmarshes and they are important for the production of organic material and nutrient cycling (Fagherazzi *et al.*, 2012). Soil organic matter (SOM) consists of animal, living plant, and microbial biomass, as well as decaying plants and humus, which is formed through microbial activity (Cambardella, 2005). SOM helps retain water and nutrients, increases soil biodiversity, and benefits soil structure (Lal, 2009), and soil has the capacity to store three times as much Carbon in SOM as in living plants or the atmosphere (Schmidt *et al.*, 2011). Research by Mossman, Davy and Grant (2012) showed that SOM was significantly less on MR sites than on accidently realigned sites and reference marshes, which may have an impact on some of the ecosystem services that MR sites can provide.

Saltmarshes are vulnerable to the effects of long-term climate change (Fagherazzi *et al.*, 2012), but they have the capacity to store carbon and only release minor levels of greenhouse gasses (Chmura *et al.*, 2003). The potential for saltmarshes to be significant

carbon sinks is recognised by many (e.g. Turner *et al.*, 2007; Cai, 2011; Callaway *et al.*, 2012; Ouyang and Lee, 2013), and there is evidence that tidal saline wetlands appropriate ten times more carbon in their soil per unit area than terrestrial peatlands, without releasing significant levels of greenhouse gasses (Moffett *et al.*, 2010). It is suggested that wetlands are suitable for carbon storage because saturated soils have depleted levels of oxygen which, in turn, constrains microbial decomposition, thus contributing to accumulating SOM (Keller *et al.*, 2012), and plant growth in saltmarshes is responsible for carbon dioxide fixation from the atmosphere (Chmura, 2013). However, before direct comparisons between studies can be made, an important distinction must be made in relation to sediment type. There are two main types of saltmarsh sediment, organogenic, which is derived from living organisms, and minerogenic, which consists of mineral material. Saltmarshes on the eastern and southern coasts of the USA are mainly organogenic, whereas minerogenic saltmarshes predominate in North-West Europe (Hughes and Paramor, 2004). The saltmarshes of Great Britain are mainly minerogenic, which means they are formed by sandy to clayey silt rather than an organic matter supply (Allen and Pye, 1992). This may influence the comparison of rates of carbon sequestration across study sites with different sediment type, especially because organogenic sediments tend to store more carbon (Middelburg *et al.*, 1997). In the UK, carbon sequestration on MR sites has been researched, and there is evidence to suggest that such sites can sequester carbon (e.g. Adams, Andrews and Jickells, 2012), but the findings of a particular study at the Tollesbury (Essex) MR scheme, suggest that, although biogeochemical functioning will occur, it could take 100 years for MR sites to accumulate the same amount of carbon that is presently stored in natural reference marshes (Burden *et al.*, 2013).

Evidence suggests that saltmarsh systems can be sinks for nitrogen and phosphorus (Turner *et al.*, 2007; Adams, Andrews and Jickells, 2012), which are the two main macronutrients that facilitate and regulate plant growth on coastal marshes (Weis and Butler, 2009). They are important for the functionality and productivity of saltmarshes, and play a key role in nutrient transformations and carbon mineralisation (Marton and Roberts, 2014). Increased nitrogen leads to a large biomass of detritus-feeding invertebrates on saltmarshes, and it influences the morphology of plants, such as *Spartina spp.* (Valiela and Teal, 1979). However, nitrogen and phosphorus can also contribute to the eutrophication of coastal waters (Marton and Roberts, 2014; Velinsky *et al.*, 2017), so it is important to maintain, create or restore saltmarshes to ensure that this essential role of filtering excess nutrients can continue. Previous research has suggested that ecosystem processes on saltmarshes remove nitrogen through denitrification (Sousa *et al.*, 2012; Bai *et al.*, 2017; Bilkovic *et al.*, 2017; Velinsky

et al., 2017), plant uptake (Brin *et al.*, 2010; Nelson and Zavaleta, 2012) and burial in saltmarsh sediments (Adams, Andrews and Jickells, 2012). Denitrification through microbial action can remove nitrogen from the system and release it into the atmosphere before it enters estuarine waters, which reduces eutrophication (Koop-jakobsen and Giblin, 2009; Vieillard and Fulweiler, 2012), although this can lead to increased atmospheric nitrous oxide (N_2O) which is a potent greenhouse gas (Roughan *et al.*, 2018). In wetlands, phosphorus is chiefly removed via soil accretion, soil absorption, microbial action, and plant uptake (Vymazal, 2018). Soil pH is also important to consider on MR sites, because it is a key factor that influences the bioavailability and uptake of these macronutrients (Hinsinger *et al.*, 2003).

Due to anaerobic conditions, nitrogen on saltmarshes is mainly in the form of ammonium (NH_4+), but Adam (1990) suggests that nitrate (NO_3-) is also likely to be a major source of available nitrogen due to aerobic microsites, or because rhizosphere oxidation moderates the prevalence of NH_4+. Increased levels of NO_3- is also the main indicator of excess nitrogen (Fenn *et al.*, 1998; Drake *et al.*, 2008), which leads to pollution. Therefore, levels of nitrate-nitrogen (NO_3-N) entering restored or created marshes should be monitored along with ammonium-nitrogen (NH_4-N). Excess phosphate can also cause environmental degradation of coastal systems, so this should also be considered after restoration, especially because high levels of phosphate in previously diked sites can be mobilised due to the arrival of tidal water (Weis and Butler, 2009).

Salinity and waterlogging of saltmarsh soils are major factors that affect species' distribution on saltmarshes (Adam, 1990). Salinity is important to monitor during saltmarsh development because high salinities may inhibit the germination and seedling establishment of saltmarsh species, and low salinities may provide the conditions for glycophytes (low salinity halophyte species) to flourish and exclude halophytic vegetation (Wolters, Garbutt and Bakker, 2005). There is evidence to suggest that saltmarsh plant species would establish in low salinities if it were not for intense interspecific competition, but as such they are restricted to high-salinity locations (Crain *et al.*, 2004). Salinity levels on saltmarshes can also have an influence on the presence and abundance of invertebrates such as beetles (Pétillon *et al.*, 2008). However, it has been argued that soil salinity alone is not a reliable parameter to explain the spatial distribution of halophytic plant species (Silvestri, Defina and Marani, 2005), with soil moisture also playing a role.

Waterlogging puts plants under stress and can lead to reduced growth due to oxygen deficiency (hypoxia) or anoxic conditions (Irfan *et al.*, 2010). Soil compaction can lead to

waterlogging (Batey and McKenzie, 2006), because it affects porosity and hydraulic properties (Mossadeghi-Björklund *et al.*, 2016), and there is evidence to suggest that poor drainage can limit the anticipated benefits of saltmarsh restoration (Spencer *et al.*, 2017). Compaction can also decrease root penetration in soils (Nawaz, Bourri and Troland, 2013), which could potentially affect vegetation succession on a saltmarsh that is reclaimed from agricultural land. As the levels of compaction from agriculture practices are apparent, it is reasonable to suggest that heavy machinery used during saltmarsh creation could have a similar affect. Evidence indicates that some saltmarsh species, such as *P. maritima*, can tolerate waterlogged soils in comparison to other saltmarsh species (Cooper, 1982), so this may be a determinate of species composition on a restored or created marsh.

1.3.3.1 Past land use

Historically, the majority of reclaimed coastal wetlands have been created on agricultural sites, such as pasture and crop-land (Junk *et al.*, 2013), hence many MR sites are being created on agricultural land. This can have an impact on saltmarsh succession on restored sites because the soil has been altered substantially by humans, in many cases over hundreds of years. Arable land would have undergone extensive tillage and different crop rotations, and pesticides would have been added. Pasture would have been used extensively by domestic grazing animals, such as cattle and sheep, and type of animal stock will have had different impacts on the habitat due to feeding behaviour and poaching (Rook *et al.*, 2004).

Different past practices can potentially have an influence on plant colonisation on an MR site. For example, arable land may have a higher pH due to the addition of lime, and it is likely that fertilizer or manure would have been added (Haynes and Naidu, 1998), which contains nutrients such as nitrogen and phosphorus. Additionally, arable systems may have contained nitrogen-fixing legumes as part of the crop rotation, which could have added to the organic matter pool and increased N mineralisation in the soil (Peoples *et al.*, 2009). The substrate on pastures may also be high in nitrogen and SOM due to animal faeces that have broken down into the soil, and evidence suggests that grassland soils, such as pasture, have higher organic matter than arable soils (Nierop, Pulleman and Marinissen, 2001; Pulleman *et al.*, 2005). Grazing intensity can also have a positive influence on the amount of nitrogen and phosphorous in grasslands (Wei *et al.*, 2011), which may impact halophytic plant succession after de-embankment on an MR site.

Both arable and grassland substrates may be compacted, the former by heavy machinery, and the latter due to heavy trampling by livestock (Hamza and Anderson, 2005),

which could affect root penetration by plants and the colonisation of benthic invertebrates on restored sites. There may be better drainage on some land than others, with certain areas periodically under the influence of standing water and other sites leaching nutrients such as phosphorus and nitrogen through surface runoff (King *et al.*, 2015). Arable and pasture sites will also have different plants on them, from crop species, to dense grasses on pastures, which may have an influence on trapping of sediment or erosion in the early stages of saltmarsh development, especially because wave action on plants can cause erosion by scouring the substrate (Bouma *et al.*, 2009). Ruderal species, such as thistles, may colonise agricultural land through disturbance and because of the influence of time between the initiation of the scheme and de-embankment; there may also be areas of bare ground due to cessation of agricultural practices.

It has been suggested that past-land use practices could cause issues with the delivery of seeds to a restored saltmarsh, and there may also be implications for plant growth and nutrient cycling due to unsuitable environmental conditions, which may in turn affect the provision of ecosystem services (Spencer and Harvey, 2012). Lawrence *et al.* (2018) found that the topography of MR sites is similar to agricultural land, and MRs are flatter and more likely to accumulate water than a natural saltmarsh. This lack of topographical heterogeneity can limit the establishment of plant communities that are typically found on saltmarshes (Brooks *et al.*, 2015). Although there are a growing number of studies based on succession on restored sites that were once reclaimed (e.g. Garbutt *et al.*, 2006; Garbutt and Wolters, 2008; Wolters *et al.*, 2008; Davy *et al.*, 2011; Mossman, Davy and Grant, 2012; Chang *et al.*, 2016; Sullivan *et al.*, 2018) few have specifically compared succession on different starting states, such as arable versus grazing pasture.

1.4 Summary

The key questions that emerge from the literature are 1) How long does it take for restored saltmarshes to develop structural equivalence to mature saltmarshes? 2) How does past land-use, such as farming practices, influence ecological development? 3) What are the impacts of restoration methods, such as the use of heavy machinery, on ecological processes?

Few long-term studies have been conducted to address these factors, which are important when considering future saltmarsh restoration projects. For example, future restoration design can be enhanced if there is a clear understanding of how the starting state influences succession of plant and animal communities. Moreover, compaction from past land-use, or restoration methods, may have a negative impact on ecological processes, thus a recognition of this issue may inform decisions that help in the rapid transformation of

restored saltmarshes into sites that are structurally equivalent to mature marshes and develop into fully functioning ecosystems that perform ecosystem services.

1.5 Steart Marshes

Steart Marshes is a working wetland site that is managed by the Wildfowl and Wetlands Trust (WWT). It is a recently recreated salt marsh (2014), providing an ideal system to understand early processes of saltmarsh development on former agricultural land. A large part of the site has been reconstructed to provide a salt marsh as part of compensatory habitat using the realignment of defence method of implementation. Extensive areas (c.300 hectares) of agricultural land were taken out of production and reconfigured to form an intertidal area with an extensive herring-bone creek system (Figure 1.5). Following reprofiling, the sea wall was breached in September 2014. The main coastal marsh site (Steart Marsh) is inundated by tidal water post breach, and it is anticipated that ecological processes will alter from the starting state to eventually form structural and functional similarities to a mature saltmarsh. WWT also manage another adjacent coastal marsh called Otterhampton Marsh, which has a management regime that differs from Steart Marsh (controlled by culverts and sluices) and is inundated by brackish water.

Figure 1.5. a) Study site at the Steart peninsula in 2010 showing recent agricultural land use; b) Study site at the Steart peninsula in 2016 showing the herring-bone creek system that was originally excavated pre-2014. The breach point is shown in the black square (Google Earth, 2018).

1.6 Aims and objectives

Saltmarsh restoration and creation is complex, and few studies have monitored ecological changes from the outset. Assessing initial changes from existing habitats is important and results from such data can facilitate decision making by project and site managers during the planning and implementation phases of MR schemes. Therefore, the aim of this project is to understand the ecological changes following reconstruction and breaching of Steart Marsh, with the primary objectives of:

1. Determining how soil, plant and invertebrate characteristics change following a breach of existing sea defences.
2. Determining how previous land-use and restoration techniques influence changes of developing saltmarsh communities.
3. Comparing these changing communities over the initial stages of saltmarsh establishment to a managed brackish coastal wetland site that was previously agricultural land and to a long-established reference (adjacent) saltmarsh community.

The expectation is that the new MR site will tend towards having community similarities with the mature saltmarsh adjacent to the reconstructed site, but changes might also be related to past agricultural use and the influences of activities associated with restoration prior to breaching. The managed brackish coastal wetland site is incorporated because it was created in the same timeframe as the MR and has a similar starting state (agricultural) to Steart Marsh. It will be of interest to assess if there are similarities between these different coastal wetland sites, or if Steart Marsh is quickly tending toward traditional saltmarsh communities as anticipated. Because of the part-time nature of this PhD programme, sampling of data relating to rates of change and convergence/divergence of community features in relation to physical factors are analysed on the site from 2014 until the latter part of 2017. This is a relatively long time-series for a study like this.

To facilitate this study, samples of the baseline community and soil characteristics that the newly constructed site had, were undertaken prior to the breaching of the sea wall to assess how the site has diverged and developed into saltmarsh from different starting states. Samples of vegetation, invertebrates and soil have been collected to analyse the progression of the newly constructed site following breaching. Importantly, a series of comparative sites have been chosen, reflecting differences in original land use and also differences in the extent of engineering (chiefly reprofiling and heavy machinery compaction) during the engineering phases pre-seawall breaching.

CHAPTER 2: STUDY SITE PREPARATION

2.1 Introduction

The preparation of study sites on Steart Marsh and Otterhampton Marsh had to be carefully considered in terms of the environmental impact of the research and health and safety. It was important to select sites that could be continually accessed post seawall breaching, within the remit of WWT that sampling and access had a low impact on flora and fauna, especially breeding birds. WWT risk assessments were used throughout the study, and specific risk assessments for equipment and methodology were created. Permission to use a SSSI saltmarsh was granted by Natural England and a separate risk assessment that was pertinent to a natural system was created and issued to the organisation. Study plots and their locations were designed specifically to satisfy the requirements of WWT, with a view to minimising impact on breeding birds.

This chapter covers the selection and preparation of study sites and plot design. Specific methodologies for soil (Chapter 3), vegetation (Chapter 4), and invertebrate (Chapter 5) characteristic changes over time are included in the indicated chapters.

2.2 Site selection and preparation

In April 2014, seven potential study areas were mapped out on Steart Marsh, hereafter labelled A-G (Figure 2.1). These plots were positioned between creek arms from the breach to the saline lagoons, and in what were separate agricultural fields prior to the breach. They were all on the northern part of the marsh because access to the southern section (below the creek system) would be problematic post-breaching.

Figure 2.1 Study areas that were proposed during the initial stages of study site preparation; a) Google Earth image post-breach showing where the plots are in relation to the creek system (original planning was conducted on paper maps pre-breach); b) Steart peninsula before site construction began, showing previous land use and field boundaries in relation to potential study plots. OM is Otterhampton Marsh and SSSI is a mature marsh under the jurisdiction of Natural England.

A consultation document written by the Environment Agency (2010) details previous land-usage, and these proposed areas were cross-referenced with this document (Table 2.1).

Table 2.1 Previous land use on each of the proposed study sites. Land-use information from the Environment Agency (2010).

Proposed study sites	Previous land-use
A	Grazing marsh pasture
B	Arable
C	Arable
D	Arable
E	Coastal grazing marsh
F	Improved grassland
G	Coastal grazing marsh

After consultation with WWT about future access to these areas it was agreed that the most appropriate areas to locate plots on Steart Marsh were Site A and Site B. These sites were likely to be the most accessible throughout the period of the study, and they were also selected because they had different starting states, with substantial heterogeneity. Site A was previously grazing marsh pasture, and Site B was previously under an arable crop rotation (Table 2.1). It was decided that it would be beneficial to locate two areas that were close to one another because they were both a similar distance from a mature saltmarsh (SSSI) that was likely to act as a seed bank. LiDAR was used to assess the elevation AOD of these sites to see how they compared to the SSSI and Otterhampton Marsh (OM). LiDAR data were obtained from the Plymouth Coastal Observatory and opened on QGIS version 2.2 to create a visual LiDAR map of the Steart peninsula. The most recent LiDAR data available in June 2014 was from 2012 before the earthworks began. The image created from the LiDAR raster tiles (Figure 2.2) was used to assess where to specifically select plots on the sites in terms of similar elevation. The image was magnified on QGIS to assess the elevations in each study site.

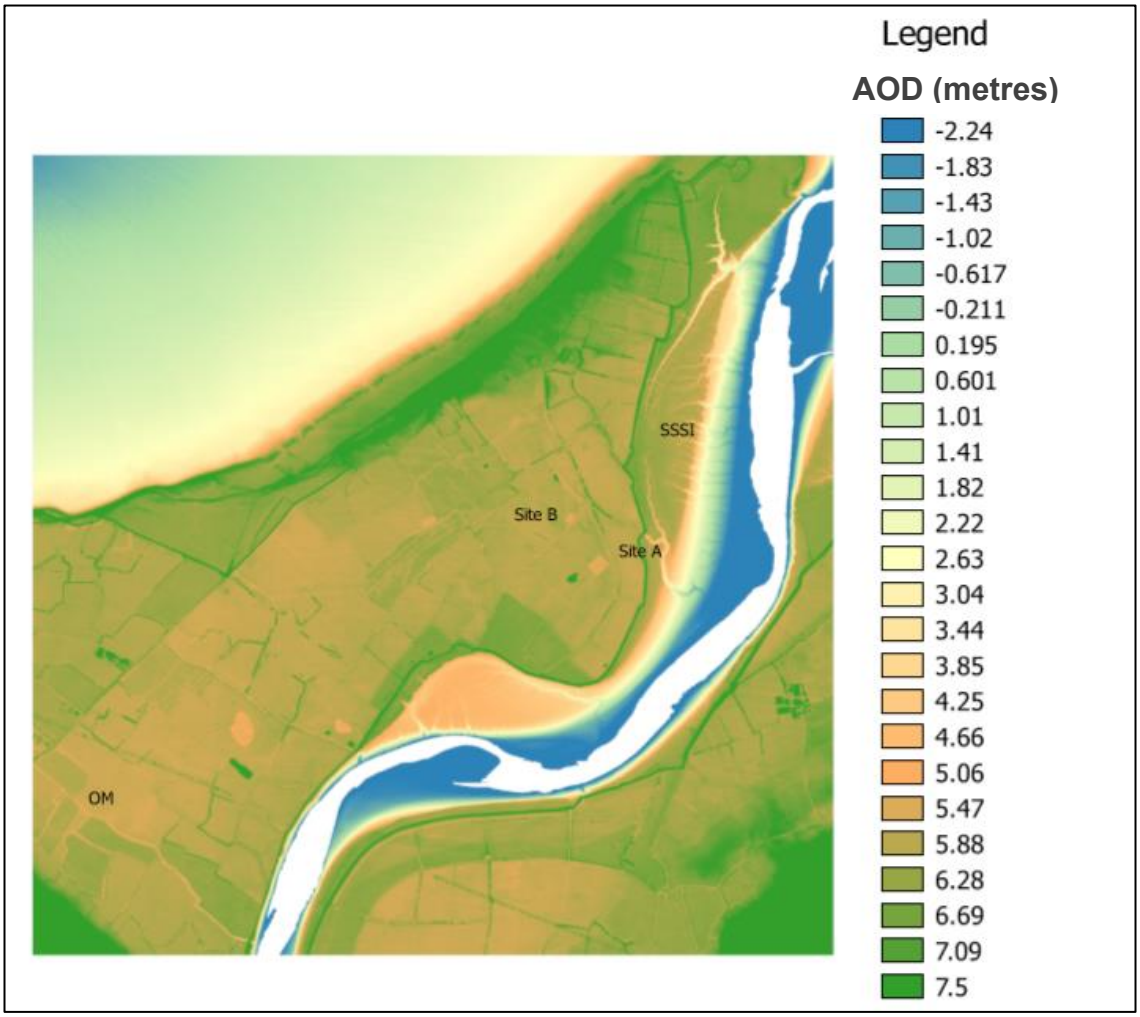


Figure 2.2 LiDAR image of the Steart peninsula in 2012. Datafiles were obtained in 2014 and originally opened on QGIS version 2.2 in June 2014. This image was created using the same datafiles on QGIS version 2.18.13.

The SSSI was specifically used as a target state and it was expected that species of vegetation found on the SSSI would colonise Steart Marsh due to their proximity. OM was ideal as a comparison to the Steart Marsh sites because it was newly created as part of the scheme and had also been agricultural land previously. It is a brackish marsh that is being managed by sluice gates, with the tides being allowed to inundate through a man-made pipeline and held in certain areas or drained depending on the needs of bird species. It is expected that succession will occur on OM, but at a different rate to that on Steart Marsh due to the management of the site, the distance from the SSSI, and the influence of brackish tidal water, but it was important to include in this study to assess if Steart Marsh plots followed a similar pattern in structural changes to OM, or if they tended toward the mature marsh as anticipated.

In summer 2014 six permanent plots were selected on Steart Marsh (Site A = 2 plots; Site B = 2 plots), OM (1 plot) and the SSSI (1 plot). It was determined that this was a preferable method to random sampling because the different starting states could be

assessed, and elevation of quadrats on the plots could be taken into account. This allowed for a comparative study of different starting states under similar levels of immersion. However, it was difficult to find exact comparative elevation AOD over such a large geographical area, and topography of the fields was inconsistent due to past land use and earthworks during creation, so the selected plots were positioned within an average of 0.5 metres in AOD of one another. A Runner 24 automatic level was used to determine accurate elevations of the selected sites. These elevations were initially taken from the nearest ordnance datum and new datums were fixed near to the study sites. This allowed for future measurements to be taken.

Although each site was categorised as grazing marsh pasture (Site A), arable (Site B and OM) and saltmarsh (SSSI), the starting states of Site A, B and OM contained ruderal species, because agricultural practices had ceased before site creation began in 2012, and the impact of earthworks on the sites was clear. Site A was more vegetated than the other sites (apart from the SSSI), and contained dense stands of terrestrial grasses along with ruderal species such as thistles; Site B and OM had patches of bare ground, ruderal vegetation, small remnants of crop species, and a variety of sparse grasses that had colonised; and the SSSI contained characteristic saltmarsh species as expected (Figure 2.3).

Figure 2.3 Starting states of the study plots; a) grazing pasture (Site A); b) arable crop rotation (Site B); c) Otterhampton Marsh; d) SSSI saltmarsh.

There were also differences between sites in terms of the disturbance generated from earth-moving equipment during creation with Site B plots clearly being the most affected. The disturbance on these Site B plots consisted of large areas of sparse vegetation, which was potentially due to compaction. Although there was some evidence of disturbance on Site A, this was minimal in comparison.

Two sample areas were positioned in Site A (A1 and A2) and two were sited in Site B (B1 and B2; Figure 2.4). In each location, the sample areas were independent from each other (> 20 metres apart) and each contained five permanently marked 2 x 2 m quadrats (Q1-Q5), which were ≥ 15 metres apart so that they maintained independence. Each plot was marked with wooden stakes. The plots were designed in this way to minimise the impact on the habitat, and to avoid encouraging birds of prey to the study sites (a specific requirement of WWT). A Runner 24 automatic level was used to determine the elevation of each quadrat, whereby five measurements were taken: one measurement from each corner of a quadrat and one in the middle.

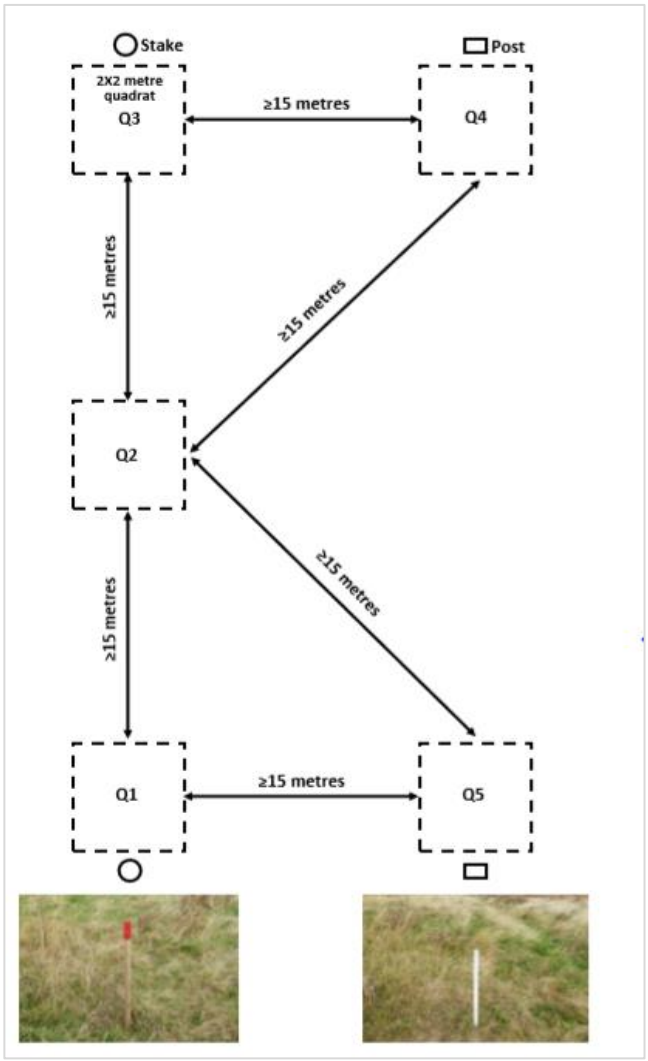


Figure 2.4 Configuration of a plot on each study site. To avoid encouraging birds of prey, the number of posts to mark plots were reduced from original plans.

In addition to the plots on Steart Marsh, one sample area was positioned on OM and one on the SSSI as a reference site (Figure 2.5).

Figure 2.5 Study plot placement on Otterhampton Marsh (OM), Steart Marsh (A1 & A2; B1 & B2) and the SSSI (Google Earth, 2018).

Downpipes were cut into sections and placed in the four corners and the middle of each quadrat on all study sites for pitfall trap placement. They were hammered into the ground so that they were flush with the surface (Figure 2.6).

Figure 2.6 Downpipe placed in the ground to provide a marker for pitfall trap placement and the corners of each quadrat. These were placed in each corner and the middle of each quadrat on all study sites.

The purpose of these pipes was to provide permanent holes where pitfall traps would be placed to catch invertebrates and to mark out the corners of each quadrat. It was anticipated that sediment would need to be removed from the top of the pipes before each sample period post-breach, but the design reduced the need for marker posts on the site that would inevitably attract birds of prey.

Invertebrate and soil characteristic data were gathered from each of these plots from 2014 (pre-breach) up to and including 2017. Accretion data were gathered in 2014 (pre-breach), 2016 and 2018, and compaction data was collected in 2018. Vegetation coverage was assessed in 2014 (pre-breach) and in each subsequent year through to and including 2018. Specific methodologies for the determination of these variables are detailed in the relevant chapter (Chapter 3: Soil Characteristics, Chapter 4: Vegetation Succession, and Chapter 5: Invertebrate Colonisation).

CHAPTER 3: SOIL CHARACTERISTICS

3.0 Introduction

Soils are the basis of functionality of saltmarshes and healthy substrates are essential for the colonisation of plants, decomposers, and macro-faunal communities.

Soil moisture and salinity are key aspects of saltmarsh ecosystems due to tidal influence. On an MR site is likely that salinity and moisture will increase after breaching in both the existing soil and in accreting sediment (Blackell *et al.*, 2004). Evidence suggests that SOC is usually high in saltmarsh soils (Turner *et al.*, 2007; Cai, 2011; Callaway *et al.*, 2012; Ouyang and Lee, 2013), so changes of this important soil variable during the early stages of succession on a new MR site are important to quantify to understand fundamental changes that may influence succession and future functioning.

Saltmarshes are generally nitrogen limited (Mcfarlin *et al.*, 2008; Gedan, Silliman and Bertness, 2009; Vivanco, Irvine and Martiny, 2015), so it was anticipated that nitrate-nitrogen and ammonium-nitrogen would drop on Steart Marsh post-breach as nitrates were expected to be initially higher on land that was formally agricultural. A similar pattern was likely to occur with phosphate due to existing high levels on farmland where fertilisers are typically added (Rhodes, 2013). It is important to assess the levels of macronutrients and pH on an MR scheme because these variables are associated with the growth and maintenance of plants (Hinsinger *et al.*, 2003), but high levels of macronutrients can also lead to pollution of waterways and estuarine systems. Additionally, SOC and nitrates in soil can lead to an increase in atmospheric greenhouse gases if they are not sequestered.

The aim of this chapter is to report and summarise key soil characteristics on Steart Marsh starting states in comparison to the SSSI saltmarsh (target state) and the managed OM study plot throughout the study period (2014 (pre-breach) to August 2017). The following variables were measured and reported in this chapter: soil water content, pH, organic matter (SOC), salinity, nitrate-nitrogen, ammonium-nitrogen and phosphate-phosphorus. Measurements of these characteristics on each study plot are reported to show how these characteristics changed in space and time over the course of the study. Principal Component Analysis and cluster analysis are used to show which quadrats on Steart Marsh are tending toward the target state, and this is shown in comparison to the managed OM study plot. These data are reported each year to assess the change over time. Changes in characteristics will help explain colonisation of vegetation and invertebrates on Steart Marsh and OM.
The subsequent sub-sections (3.1.1-3.1.7) describe the methods used to determine

measurements of the soil characteristics that were assessed in this study. These tests were carried out on all soil samples (Aug 2014-Aug 2017). The rates of accretion and soil compaction on Steart Marsh and OM are also described in this chapter.

3.1 Method

3.1.1 Soil collection method

In August each year (2014-2017) three soil cores were taken from each quadrat (five quadrats on each plot) to a depth of 20cm using a soil corer measuring 7.2cm diameter. Cores were measured and split in to 10cm sections (C1 = top 10cm and C2 = 10-20cm). The three cores from each layer were put into plastic freezer bags and sealed (one bag for each core level), taking care to remove as much air as possible. The two bags from each quadrat were labelled with plot name, quadrat number and core layer (e.g. A1 Q1 C1), and date. Bags were stored in a refrigerator in a dark room below 5° Centigrade.

3.1.2 Soil preparation and soil water content

One-hundred and fifty grams of each of the refrigerated soil samples (with invertebrates previously removed – see Chapter 5) was weighed into beakers labelled with the site name, quadrat number, core depth and date. The beaker was also weighed, and the weight recorded. Each 150g sample was air dried (below 45°C) for 72 hours and then reweighed once dry. These weights were used to calculate the air-dried water content of the soil using gravimetric measuring using the following formula:

$$WC\ (\%) = 100\ x\ \frac{weight\ before\ drying - weight\ after\ drying}{weight\ before\ drying}$$

Each sample was then ground down using a pestle and mortar and then decanted into a soil grinder (the grinder was cleaned between each sample). The ground-down, dry soil was collected into sample bags, which were labelled with site name, quadrat number, core depth and date (e.g. A1 Q1 C1, August 2014). The bags were then stored in a refrigerator (5°C) in a darkroom ready for chemical analysis.

3.1.3 pH sampling

Ten grams of each dried soil sample was weighed into 50ml beakers. 40ml of 0.01 molar $CaCl_2$ was added to each beaker, which stabilised the pH reading. A glass rod was used to mix the 1:4 soil and $CaCl_2$ solution in each labelled beaker; the rod was rinsed with deionised water between each sample to avoid contamination. The samples were left to settle for 20 minutes and then stirred again. After stirring, the samples were left to stand for another

30 minutes. A two-point calibrated Jenway 350 pH Meter was used to determine the pH of each sample. The pH meter probe was rinsed with deionised water between each sample.

3.1.4 Salinity - electrical conductivity (EC)

Electrical Conductivity was used as a proxy for salinity. A Palintest Multiparameter Pocket Sensor was used to determine EC of each sample. A 1:5 dilution (1-part dry soil: 5 parts deionised water) was used to determine EC in each sample, and Palintest Multiparameter Pocket Sensor instructions for testing soil $EC_{1:5}$ were followed. The Pocket Sensor was calibrated using mid-range and high-range conductivity standards. A level 10ml scoop of dry soil was added to a clean plastic container and 50ml of deionised water poured into the container. A lid was screwed tightly onto the container and the solution was shaken vigorously for two minutes. Each sample was allowed to settle before the probe was inserted, taking care to avoid covering the probe in sediment at the bottom of the container. The probe was operated at 25°C and was cleaned with deionised water between each sample. The $EC_{1:5}$ of each sample was recorded as µS/cm and converted to dS/m for analysis.

3.1.5 Soil organic matter (SOM) and soil organic carbon (SOC)

Five grams of dried soil was weighed into crucibles of known weight. The 5 gram samples were oven dried at 110°C for 24 hours or until a constant weight was achieved. Keller and Medvedeff (2016) recommend that mineral wetland soils are dried at this temperature for the analysis of SOM. Once dried, the samples were weighed using an analytical balance with a draught shield. The samples were then placed in a muffle furnace at a temperature of 430°C for six hours to ignite organic matter as used by Leelamanie, Liyanage and Rajarathna (2015). It is suggested that temperatures above 440°C will trigger the loss of structural water from clay lattices, thus overestimating SOM, but temperatures lower than 400°C may cause incomplete burning of organic matter (Leelamanie, Liyanage and Rajarathna, 2015; NRM Laboratories, 2015). The muffle furnace was turned off after 6 hours and allowed to cool before samples were removed. The ignited samples were weighed again on the analytical balance. The SOM content was determined by calculating the loss of weight between the oven-dried and ignited samples. % SOM was calculated from SOM per gram and was converted to % SOC by multiplying % SOM values by 0.58 (Bemmelen index for converting organic matter to organic carbon). % SOC was used in statistical analysis.

3.1.5.1 Bulk density calculation for SOC estimation by hectare

Bulk density was calculated from the dry soil sample used for soil moisture analysis and the known weight and volume of original soil cores from each quadrat. There was little

variation in soil moisture between quadrats each year, so this method was deemed suitable to estimate bulk density within cores from the same quadrat. The % dry weight of the 150g wet sample was calculated and used to approximate the dry weight of the three soil cores in each sample bag. This approximate dry weight and the combined volume and depth of the three cores was used in the following equation to estimate bulk density in each core layer (C1 and C2) of each quadrat:

$$Bulk\ density\ (g/cm^3) = \frac{\text{Dry soil weight (g)}}{\text{Soil volume (cm}^3\text{)}}$$

The approximate bulk density was used in the following equation to estimate SOC as tC ha $^{-1}$:

$$SOC\ tC\ ha^{-1} = \left(\frac{\%\text{SOC}}{100}\right) \text{ x (bulk density (g/cm}^3\text{) x core depth(m) x hectare)}$$

Where t is tonne, C is carbon, and ha is hectare (10,000m^2).

3.1.6 Nitrogen and phosphorus

A Palintest Professional Soil Testing Kit (model SKW400) was used to determine nitrate-nitrogen (NO_3-N), ammonium-nitrogen (NH_4-N) and phosphate-phosphorus (PO_4-P) in each soil sample. Samples were prepared and analysed according to Palintest SKW400 (NO_3-N and PO_4-P) and SKW500 (NH_4-N) instructions. A Palintest Soiltest 10 Photometer was used to determine measurements based on colorimetry, and it was calibrated regularly using supplied colour standards. All glassware was acid washed using 10% hydrochloric acid.

For NO_3-N determination in each sample, 50 ml of deionised water was measured into a 50 ml flask. One level scoop of Extract N (1 molar NH_4Cl) was added using the provided Extract N scoop. The flask was capped with a bung and shaken to dissolve the Extract N powder. A level 2ml scoop of soil was added to the solution, and the flask was shaken for 1 minute. One level scoop of Nitratest powder was added to reduce nitrate to nitrite, and the flask was shaken for another minute. Whatman 40 filter paper was used to filter the solution in to another 50 ml flask. The extraction filtrate was poured into Palintest Photometer cuvettes up to the 10 ml line, and a Nitrocol reagent tablet was added to the cuvette and crushed to achieve a colour change in the sample. The cuvettes for each sample were placed in a rack and left to stand for 10 minutes so that a complete colour change could occur. Phot 007 was selected on the Soiltest 10 Photometer for NO_3-N and the Photometer was blanked using clear extract (before the Nitrocol reagent tablet was added as per the

instructions). Each sample was placed in the Photometer and covered with the supplied cap to prevent external light affecting the results. The NO_3-N readings were recorded in mg/l and converted to parts per million (ppm). Many of the NO_3-N readings were above the range of 0-25 mg/l so it was decided that all samples would be diluted to 1:5 using volumetric flasks. This allowed for comparability across all samples. However, three of the 2014 samples had to be diluted further due to extremely high NO_3-N compared to other samples. These were A1 Q2 C1, A1 Q3 C1 and A2 Q4 C1 which were diluted to 1:20.

NH_4-N and P extraction followed similar procedures. For NH_4-N determination, 50ml of deionised water was added to a flask and five Extract A (1 molar KCl) tablets were added and dissolved. One 10 ml scoop of soil was added to the solution and shook for two minutes. The solution was filtered into another flask using Whatman 40 filter paper. Exactly 1 ml was taken from the extraction filtrate using a syringe and placed in a clean cuvette. The cuvette was topped up to the 10 ml mark with deionised water and one Ammonia A No 1 tablet and one Ammonia S No 2 tablet were added and crushed simultaneously until fully dissolved. The cuvettes were left to settle in a rack for 15 minutes until full colour development was achieved. Phot 002 was selected on the Soiltest Photometer and a blank was used before sample readings were taken. NH_4-N readings were recorded in mg/l and converted to ppm. After trialling this method with a range of the soil samples it was determined that a 1:5 dilution was needed because most samples were above the standard range of 0-75 mg/l. The extract of each sample was diluted in 50ml volumetric flasks prior to adding the reagent tablets. The NH_4-N test was carried out on the 1:5 diluted samples.

For PO_4-P determination, 50ml of deionised water was added to a 50 ml flask and five Extract P (0.5 molar $NaHCO_3$) tablets were added and dissolved. A level 2ml scoop of soil was added and shaken for one minute. The solution was filtered into another flask using Whatman 40 filter paper. Exactly 2 ml was taken from the extraction filtrate using a syringe and placed in a clean cuvette. The cuvette was topped up to the 10 ml mark with deionised water and one Acidifying S tablet was added. This was crushed and mixed until it dissolved. One Phosphate P tablet was added, and it was crushed until it dissolved. The cuvettes were placed in a rack for ten minutes until the colour had developed. Phot 008 was selected on the Soiltest Photometer and a blank was used before phosphate readings were taken. PO_4-P readings were recorded in mg/l and converted to ppm. This method is based on Olsen's Method (1982) of soil P determination.

3.1.7 Accretion

Originally, accretion data was gathered by measuring two posts on each of the sample plots in the summer each year post-breach. However, due to concerns with the posts on the sites being damaged by grazing animals, an automatic level was used in 2016 to gain accurate elevation data of each quadrat. This was directly comparable to data collected prior to the breach in September 2014, because levels of each quadrat were taken to ensure comparability of plot height. Final elevations on the plots were taken in August 2018 to assess if the rate of accretion had changed after final samples were taken in 2017. The site was visited in the autumn and winter of 2017 to gather accretion data, but the study sites could not be accessed due to heavy rain affecting the solidity of substrate and dredged material blocking safe passage to the plots. Therefore, final elevations on the plots were taken in August 2018 to assess if the rate of accretion had changed after final samples were taken in 2017.

3.1.8 Compaction

During the study, compaction was highlighted as a potential issue affecting succession. This was not initially quantified, but compaction data were collected in August 2018 to assess if there were differences between the plots. A Dickey-John soil compaction tester was used to collect these data. According to operating instructions, moderate compaction is > 200 pound-force per square inch (psi) and high soil compaction is > 300psi, so these values were used to assess compaction levels. Five samples were taken from each quadrat (one from each corner and one from the middle).

3.1.9 Statistical analysis

In this chapter measurements of soil characteristics on each plot were compared and differences in these variables between years were assessed (2014 - 2017). Cluster analysis was used to show similarities between plot quadrats in relation to soil characteristics in C1 and C2 cores each year (2014-2017) and PCA analysis was used to show how the quadrats were ordinated in each year of the study in relation to the soil characteristics in each core level (C1 and C2). Procrustes Error plots were used to show how the quadrats within plots changed in relation to each other from August 2014 to August 2017. Minitab 18 was used for all one-way ANOVA analyses (for those data sets that met the assumptions of the test) and subsequent post-hoc tests to determine which plots were different when differences between plots were detected with the ANOVA. This statistical package was also used for cluster analysis and descriptive data. R version 3.5 was used for Kruskal-Wallis tests for the data sets that did not meet the assumptions of ANOVA, along with subsequent post-hoc tests

using the PMCMRplus package (Pohlert, 2018). PCA ordination and Procrustes error plots were carried out on the Vegan package (Oksanen *et al*., 2017)..

3.2 Results

3.2.1 August 2014 (Pre-breach baseline data) – differences between plots

In August 2014 there were notable differences between most of the soil characteristics of the SSSI sample plot and the Steart Marsh and Otterhampton Marsh plots. This was true of both C1 and C2 layers (Table 3.1). This was expected due to the difference in starting states of each plot. However, significant differences were not detected in all instances between the SSSI and Steart Marsh and OM plots, and there was considerable variation in nitrate-nitrogen levels across the plots (Table 3.1).

Table 3.1 Mean (± Standard Error of the Mean) of soil variables in each soil core (C1 and C2) pre-breach (August 2014). *SSSI data are from August 2016 and are included for comparison.

August 2014 C1	**A1**		**A2**		**B1**		**B2**		**OM**		**SSSI***	
	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM
Moisture (%)	17.44	1.42	14.01	2.98	13.84	1.93	20.83	1.04	24.57	0.43	38.10	1.56
pH	7.34	0.16	7.17	0.21	7.11	0.12	7.27	0.14	7.25	0.10	7.37	0.03
EC (dS/m)	0.48	0.10	0.39	0.09	0.16	0.03	0.16	0.02	0.31	0.03	6.10	0.08
SOC (%)	5.37	1.38	4.76	1.46	3.37	0.29	3.55	0.13	3.64	0.13	6.92	0.40
NO_3-N (PPM)	125.40	62.23	101.80	47.51	17.00	12.16	2.90	1.24	0.70	0.44	3.60	3.12
NH_4-N (PPM)	128.00	35.68	146.50	50.38	73.00	11.39	111.50	18.19	57.00	7.76	99.50	11.02
PO_3-P (PPM)	9.80	1.24	12.00	4.00	24.00	2.70	27.60	1.60	14.00	2.00	23.00	3.35
August 2014 C2	**A1**		**A2**		**B1**		**B2**		**OM**		**SSSI***	
	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM
Moisture (%)	17.94	3.43	19.19	1.05	18.61	1.03	19.58	2.34	24.60	1.04	33.91	1.67
pH	7.45	0.04	7.31	0.16	7.33	0.04	7.19	0.10	7.33	0.09	7.37	0.03
EC (dS/m)	0.31	0.05	0.22	0.05	0.14	0.01	0.21	0.05	0.47	0.06	5.33	0.31
SOC (%)	3.51	0.73	3.32	0.74	3.01	0.44	3.44	0.19	3.28	0.17	6.43	0.27
NO_3-N (PPM)	30.20	26.26	32.40	31.41	1.00	0.69	29.60	23.86	1.20	0.64	4.20	1.96
NH_4-N (PPM)	76.50	10.48	145.60	42.61	66.00	3.22	70.00	6.37	47.50	4.47	72.00	8.38
PO_3-P (PPM)	16.60	2.98	11.00	2.98	26.00	6.02	22.40	3.14	12.80	1.59	18.20	1.69

3.2.1.1 Soil moisture

Soil moisture was higher on the SSSI (M=38.10%, SEM ± 1.56) than on the other plots in 2014 (Table 3.1 and Figure 3.1).

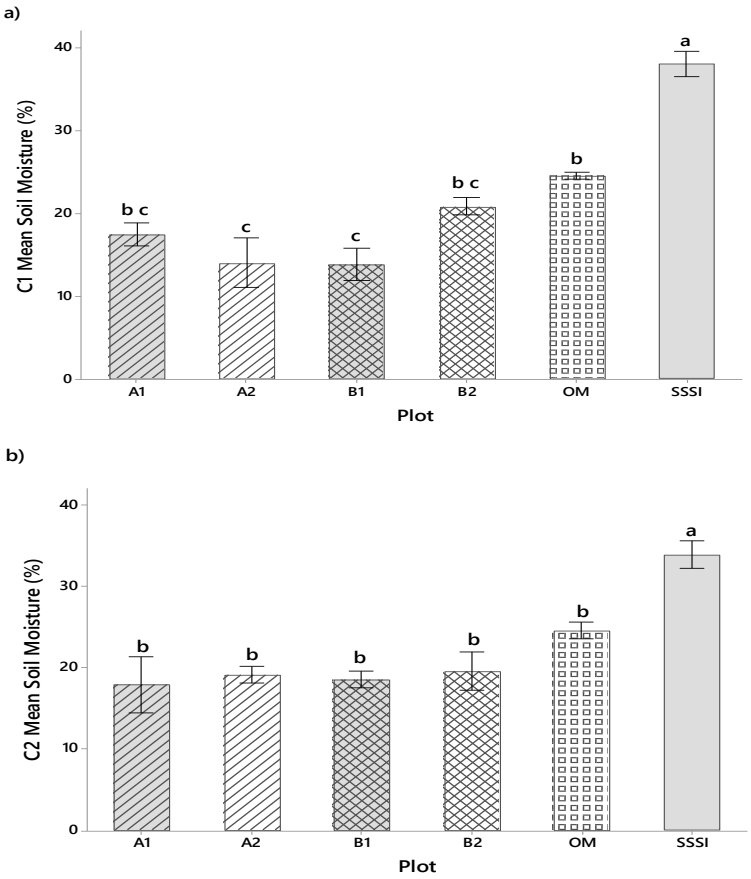


Figure 3.1 Average % soil moisture on each plot in August 2014. SSSI data are from 2016; a) % soil moisture in C1 (top 10cm soil) (n = 5 in each plot); and b) % soil moisture in C2 (10-20 cm depth) (n = 5). Error bars represent standard error of the mean. Means that do not share a letter are significantly different ($p < 0.05$, determined by ANOVA).

The highest average moisture content of soil on the other plots was on OM (M = 24.57%, SEM ± 0.43). This may be because the sluice that feeds tidal waters to OM was opened in July 2014 (Laver, 2016), two months before the breach was completed on Steart Marsh. A similar result was found in the second layer of soil (C2) (Table 3.1). There was a significant difference between the % moisture of C1 in the plots (one-way ANOVA, $F(5, 24) = 27.26$, $p < 0.001$), and a Tukey post-hoc showed there was a significant difference between moisture % on the SSSI and all other plots. Moisture % on OM was significantly different from A2 and B1, but not A1 and B2. A difference in C1 % moisture was not detected between the A and B plots in August 2014 (Figure 3.1a). In C2 cores there was also a significant difference between plots in 2014 (one-way ANOVA, $F(5, 24) = 9.78$, $p < 0.001$), and the difference in moisture % was between the SSSI and all other plots. No differences in moisture % were detected between the A, B and OM plots in C2 cores (Figure3.1b).

3.2.1.2 pH

pH readings were similar across all plots. The average starting state soil across Steart Marsh and Otterhampton Marsh was alkaline (> 7.0). This was marginally higher than expected because, according to the Natural Environment Research Council (2015), the topsoil in this area had a pH between 6.5 and 7.2 pre-breach. Some samples were > 7.2, but none of the samples fell below 6.5. The SSSI also had alkaline soil in C1 and C2 cores (both M = 7.37 ± 0.03). It was determined that that there was no significant difference between the pH of C1 cores between plots (one-way ANOVA, F (5, 24) = 0.51, p = 0.763) or between the pH of C2 cores between plots (one-way ANOVA, F (5, 24) = 0.90, p = 0.500) (Figure 3.2).

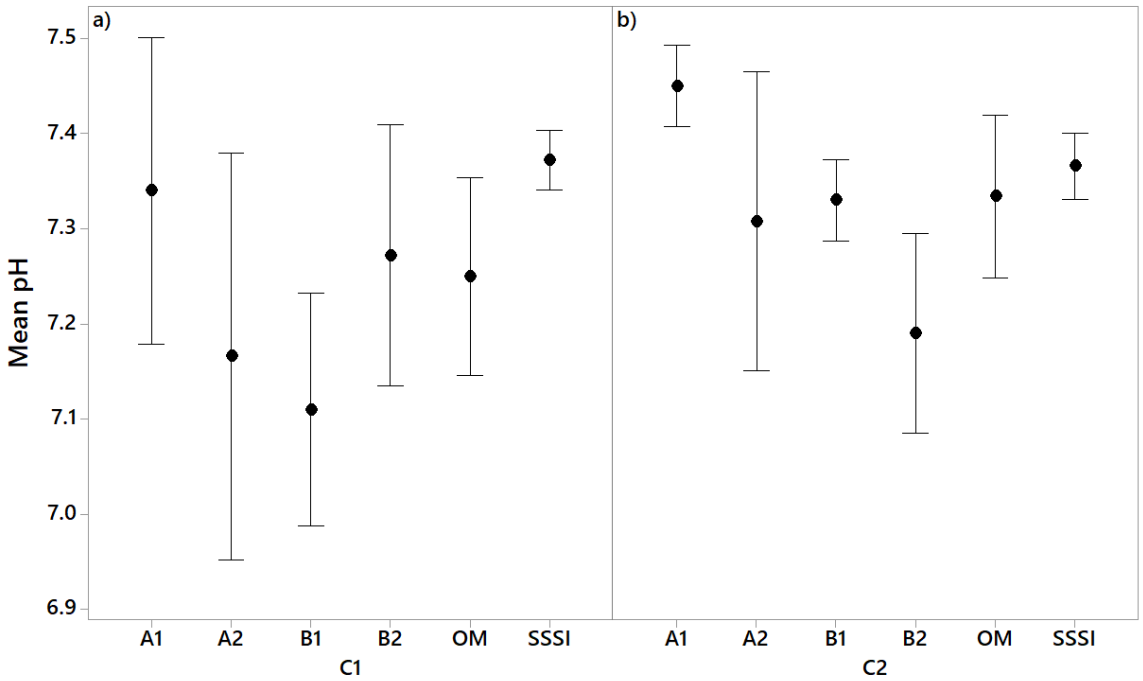


Figure 3.2 Mean pH on each plot in August 2014. SSSI data are from 2016; a) Mean pH in C1 (top 10cm soil) (n = 5 in each plot); b) Mean pH in C2 (10-20 cm depth) (n = 5). Error bars represent standard error of the mean.

3.2.1.3 Electrical conductivity

In 2016 the C1 SSSI samples had a mean value of 6.10 dS/m. Pre-breach, A1 had the highest average mean EC of the terrestrial plots (0.48 dS/m) and Site B plots had the lowest average EC (Both 0.16 dS/m). A Kruskal-Wallis showed that the EC in the C1 cores was significantly different between the plots (Kruskal-Wallis, H (5) = 20.65, p = 0.001). Despite the high level of EC in the SSSI, post-hoc Dunn pairwise comparisons tests with Holm correction did not detect a difference between the Site A plots, OM and the SSSI. The SSSI had significantly higher EC than the Site B plots. The EC in the C2 cores was also

significantly different between plots (Kruskal-Wallis, H (5) = 21.65, P = 0.001). Dunn pairwise comparisons with Holm correction showed that EC in A2, B1 and B2 was significantly lower than in the SSSI, but a difference was not detected between A1, OM and the SSSI (Figure 3.3b).

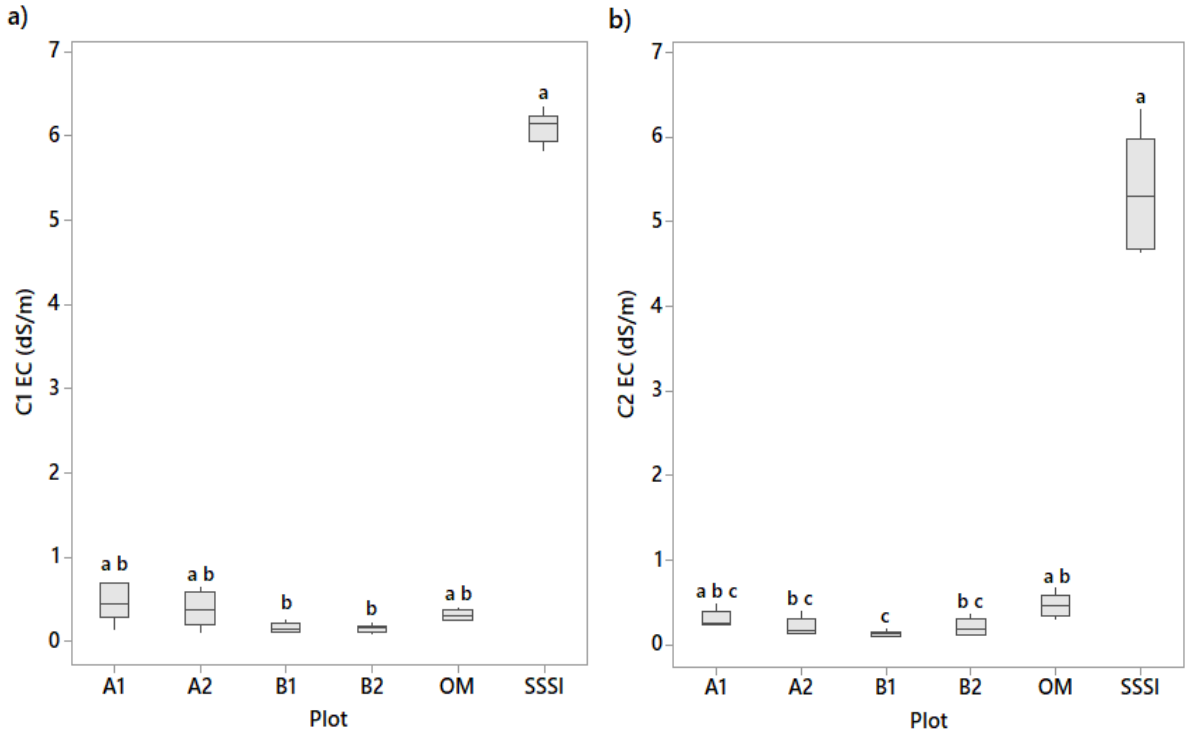


Figure 3.3 Median EC of a) C1 cores (n = 5 in each plot) and b) C2 cores in each plot (n = 5) in 2014. SSSI data are from August 2016.

3.2.1.4 Soil organic carbon

In August 2014, the top 10cm of soil (C1) on the pasture plots, A1 and A2, had an average % soil organic carbon (SOC) of 5.37 ± 1.38 (SEM) and 4.76 ± 1.46 respectively. The arable plots on Steart Marsh, B1 and B2, had an average % SOC of 3.37 ± 0.29 and 03.55 ± 0.13 respectively, and the OM plot had an average % SOC of 3.64 ± 0.13. In comparison, the SSSI topsoil had an average % SOC of 6.92 ± 0.40 (Table 3.1). C1 data were non-normal, so a Box-Cox transformation was conducted. Transformed data were also non-normal, so a Kruskal-Wallis test was used to compare plots. The test showed that there was not a significant difference of % SOC in C1 between plots (Kruskal-Wallis, H (5) = 6.96, p = 0.224).

In C2 cores, A1 and A2 had an average % SOC of 3.51 ± 0.73 and 3.32 ± 0.74 respectively. B1 and B2 had an average % SOC of 3.01 ± 0.44 and 3.44 ± 0.19 respectively, and the OM plot had an average % SOC of 3.28 ± 0.17. The C2 cores on the SSSI had an

average % SOC of 6.43 ± 0.27 (Table 3.1). C2 data were transformed (Box-Cox transformation) and were found to be normally distributed. It was determined that there was a significant difference between plots (one-way ANOVA, $F(5, 24) = 6.99$, $p < 0.001$), and a Tukey pairwise comparison showed that the % SOC in the SSSI C2 cores was different than all other plots (Figure 3.4). No difference was detected between the Steart Marsh and OM plots.

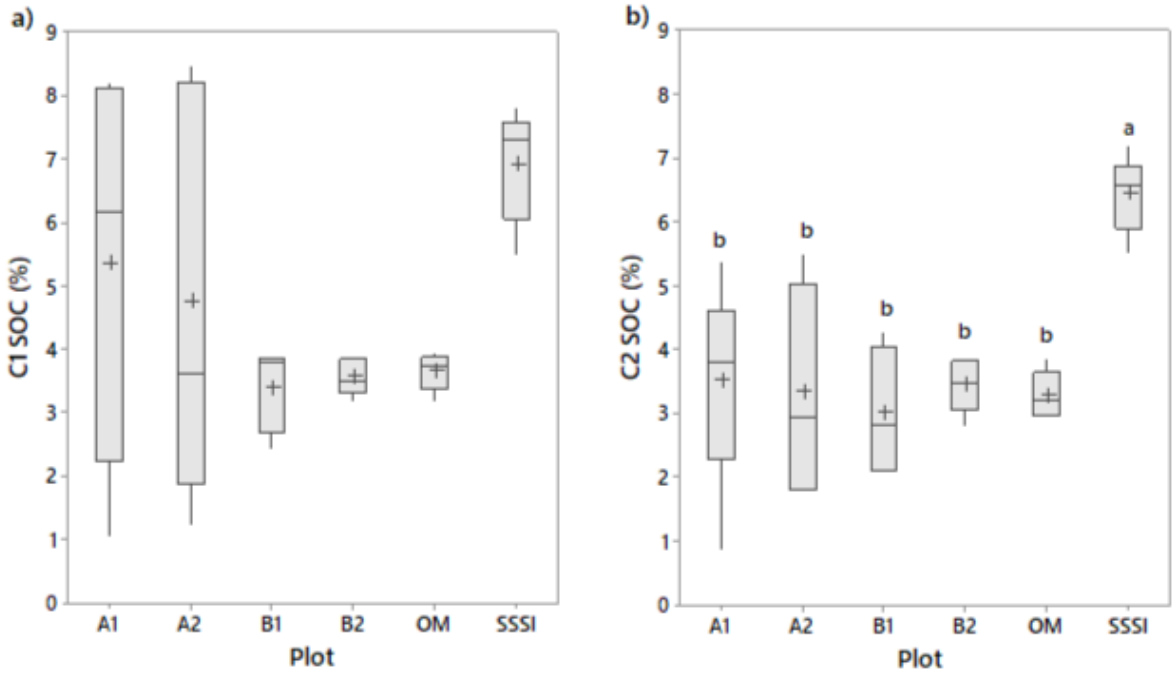


Figure 3.4 Median % SOC of a) C1 cores (n = 5 in each plot) and b) C2 cores in each plot (n = 5) in 2014; crosses (+) represent the mean of % SOC in each plot and means in b) that do not share a letter are significantly different ($p < 0.05$). SSSI data are from August 2016.

3.2.1.5 Macronutrients

NO_3-N levels in August 2015 were variable, especially on the Site A plots (Figure 3.5a and 3.5b). Maximum levels in the soil cores in A1 and A2 were 280 ppm and 236 ppm respectively. However, NO_3-N was not detected in some C1 samples. Levels were generally lower in C2 cores, but there was also variation in these samples ranging from 0-158 ppm on the Site A plots. B1 and B2 had a much lower average and median level of NO_3-N than the pasture sites, but there were some high levels detected in the C2 cores, especially in quadrat B2 Q3 C2 which was 124 ppm. OM had very low levels of nitrates in C1 (0-2 ppm) and C2 (0-3 ppm) cores. In comparison, there were relatively low levels of NO_3-N detected on the SSSI in 2016. Levels ranged from 0-16 ppm in C1 and 0-10 ppm in C2 cores. Data were non-normal after transformation (Box-Cox) so a Kruskal-Wallis test was used to compare NO_3-N levels in plots. There was not a significant difference between nitrate levels in C1

cores between plots (Kruskal-Wallis, H (5) = 6.83, p = 0.233 adjusted for ties) or in C2 cores between plots (Kruskal-Wallis, H (5) = 4.16, p = 0.526 adjusted for ties).

NH_4-N levels were also variable. Ammonium levels were high on the SSSI in comparison to NO_3-N as expected. Site A plots were more variable than the Site B plots and OM. NH_4-N levels were higher on average than NO_3-N levels in all plots. A one-way ANOVA was conducted after Box-Cox transformation, and it did not detect a significant difference in ammonium levels between plots in C1 cores (one-way ANOVA, F (5, 24) = 1.53, p = 0.219) (Figure 3.5c), but there was a difference detected between plots in C2 cores (one-way ANOVA, F (5, 24) = 5.47, p = 0.002). A Tukey pairwise comparison showed that plots A1 and A2 were significantly different to OM, but no differences were detected between all other plots (Figure 3.5d).

The highest phosphate levels were found in the Site B plots and the SSSI. There was a significant difference between plots in C1 cores (one-way ANOVA, F (5, 24) = 7.65, p <0.001) and between plots in C2 cores (one-way ANOVA, F (5, 24) = 2.79, p = 0.040). A Tukey pairwise comparison showed that in C1 cores phosphate levels in A1 and A2 plots were different than B1 and B2 plots and B2 was different than OM (Figure 3.5e). A difference was not detected between the other plots. In C2 cores phosphate levels in A2 were different to those in B1, but no difference was detected between the other plots (Figure 3.5f).

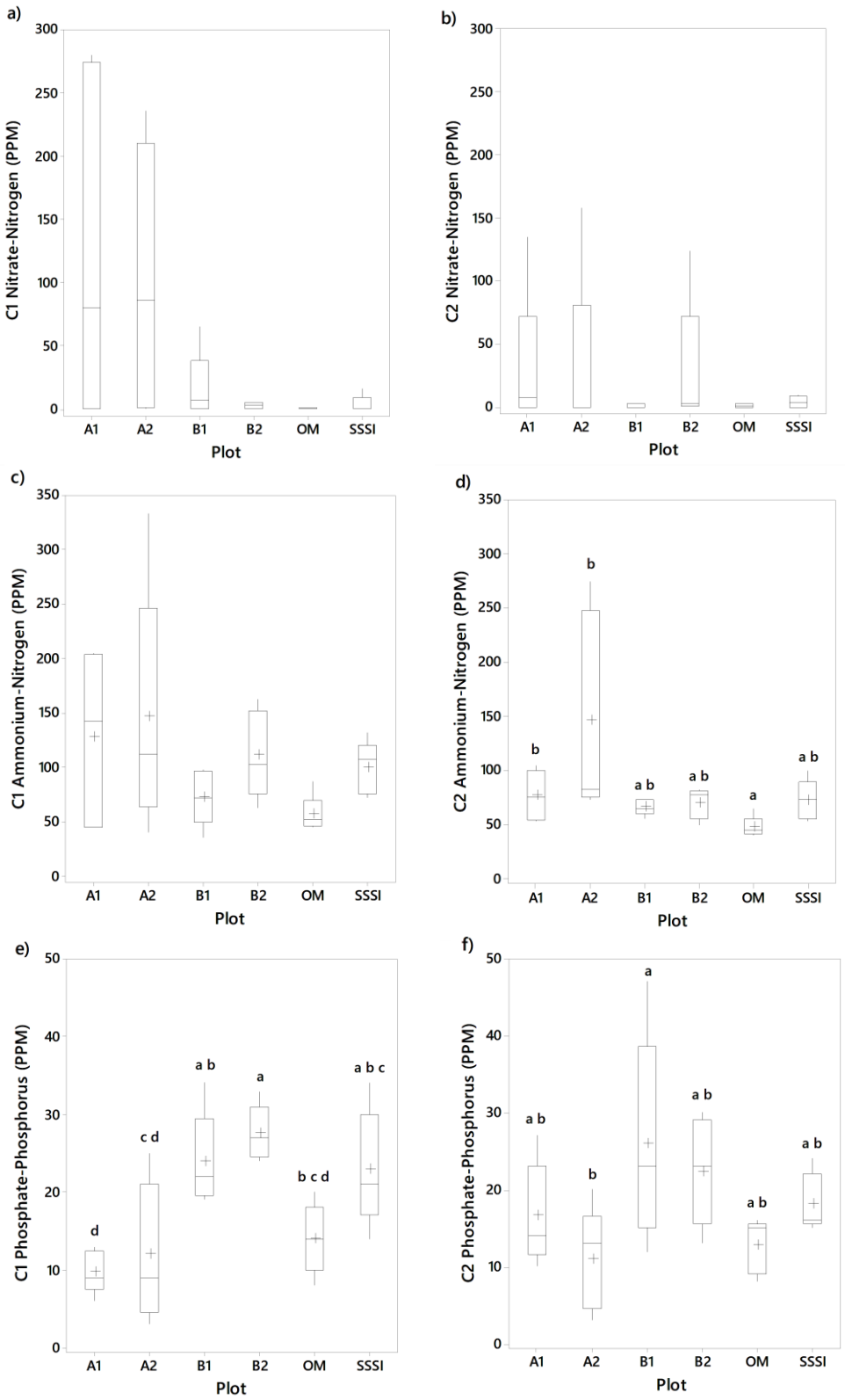


Figure 3.5 Average macronutrients each year (2015-2017); a) Nitrate-Nitrogen in C1 cores (n = 5 in each plot); b) Nitrate-Nitrogen in C2 cores (n = 5); c) Ammonium-Nitrogen in C1 cores (n = 5); d) Ammonium-Nitrogen in C2 cores (n = 5); e) Phosphate-Phosphorus in C1 cores (n = 5); f) Phosphate-Phosphorus in C2 cores (n = 5). Horizontal lines are medians, boxes represent the interquartile range (25-75%) and bars are minimum and maximum values. Means (+ symbol) within each boxplot that have different letters signify significant differences ($P < 0.05$). SSSI data are from August 2016.

3.2.2 Post-breach soil characteristics - differences between plots

In the years following the breach (2015-2017) there were changes in soil variables and between plots in C1 and C2 cores (Table 3.2). Each soil variable was compared between plots and to the SSSI target state.

Table 3.2 Mean (± Standard Error of the Mean) of soil variables in each soil core (C1 and C2) each year post-breach (August 2015-2017). *SSSI data are from August 2016.

August 2015 C1	**A1**		**A2**		**B1**		**B2**		**OM**		**SSSI***	
	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM
Soil Moisture (%)	28.04	0.97	30.29	2.48	19.24	1.95	21.10	1.26	21.54	2.87	38.10	1.56
pH	7.23	0.04	7.40	0.06	7.35	0.05	7.43	0.04	7.01	0.07	7.37	0.03
EC (dS/m)	4.03	0.60	3.87	0.48	3.86	0.43	4.50	0.39	1.13	0.18	6.10	0.08
SOC (%)	4.67	0.76	5.06	0.95	3.18	0.17	2.96	0.16	3.06	0.10	6.92	0.40
NO_3-N (PPM)	1.00	0.77	58.00	28.70	17.20	4.12	49.80	4.43	16.60	4.70	3.60	3.12
NH_4-N (PPM)	81.50	22.51	73.00	20.18	39.50	2.78	65.50	10.99	68.00	8.34	99.50	11.02
PO_3-P (PPM)	16.80	1.11	16.40	1.60	22.80	3.25	29.60	2.25	13.40	1.29	23.00	3.35
August 2015 C2	**A1**		**A2**		**B1**		**B2**		**OM**		**SSSI***	
	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM
Soil Moisture (%)	23.75	1.35	23.17	1.29	20.26	2.91	19.52	1.71	22.53	1.53	33.91	1.67
pH	7.19	0.05	7.32	0.06	7.12	0.05	7.29	0.05	7.03	0.09	7.37	0.03
EC (dS/m)	3.35	0.17	2.99	0.37	3.13	0.39	3.10	0.16	0.97	0.08	5.33	0.31
SOC (%)	3.98	0.56	3.80	0.76	3.17	0.21	2.63	0.20	2.89	0.10	6.43	0.27
NO_3-N (PPM)	9.30	5.17	22.90	6.52	16.40	4.40	21.00	4.22	4.90	1.99	4.20	1.96
NH_4-N (PPM)	62.00	12.13	76.50	25.89	36.00	3.22	69.50	16.80	51.50	8.12	72.00	8.38
PO_3-P (PPM)	10.40	1.81	14.20	3.73	23.20	3.43	26.00	2.30	12.60	0.40	18.20	1.69
August 2016 C1	**A1**		**A2**		**B1**		**B2**		**OM**		**SSSI***	
	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM
Soil Moisture (%)	30.78	2.29	30.69	2.86	17.37	1.95	21.72	1.64	14.04	1.11	38.10	1.56
pH	7.33	0.05	7.33	0.04	7.19	0.05	7.25	0.03	7.02	0.05	7.37	0.03
EC (dS/m)	5.27	0.36	6.32	0.51	7.10	0.65	6.57	0.73	3.36	0.19	6.10	0.08
SOC (%)	2.78	0.26	2.81	0.24	2.89	0.20	3.59	0.15	4.24	0.10	6.92	0.40
NO_3-N (PPM)	16.20	3.57	6.50	3.06	8.10	0.78	4.20	1.76	17.00	4.97	3.60	3.12
NH_4-N (PPM)	57.00	7.04	43.00	8.89	36.50	2.03	62.00	9.76	63.50	5.16	99.50	11.02
PO_3-P (PPM)	24.60	3.11	23.20	1.59	23.20	1.32	26.40	2.18	14.00	1.14	23.00	3.35
August 2016 C2	**A1**		**A2**		**B1**		**B2**		**OM**		**SSSI***	
	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM
Soil Moisture (%)	27.57	1.48	30.82	2.05	17.60	2.50	22.57	1.32	19.12	1.26	33.91	1.67
pH	7.29	0.05	7.30	0.05	7.19	0.06	7.30	0.02	7.06	0.05	7.37	0.03
EC (dS/m)	4.78	0.40	4.91	0.55	4.87	0.26	4.09	0.29	1.97	0.12	5.33	0.31
SOC (%)	4.32	1.07	3.85	0.78	3.07	0.25	3.53	0.13	4.01	0.11	6.43	0.27
NO_3-N (PPM)	40.80	16.89	15.30	6.71	5.60	0.94	3.10	0.97	1.40	0.93	4.20	1.96
NH_4-N (PPM)	71.50	10.68	57.00	12.58	31.50	5.28	55.50	4.36	56.50	4.51	72.00	8.38
PO_3-P (PPM)	13.60	2.69	17.20	3.15	23.60	3.06	29.80	2.63	13.40	1.21	18.20	1.69
August 2017 C1	**A1**		**A2**		**B1**		**B2**		**OM**		**SSSI***	
	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM
Soil Moisture (%)	36.30	1.68	35.73	1.26	29.68	1.84	30.91	0.78	21.22	0.66	38.10	1.56
pH	7.32	0.03	7.35	0.03	7.12	0.02	7.28	0.03	6.93	0.07	7.37	0.03
EC (dS/m)	4.46	0.52	6.26	0.15	5.85	0.75	5.11	0.39	4.17	0.24	6.10	0.08
SOC (%)	5.77	0.54	4.37	0.46	4.54	0.50	3.59	0.25	3.51	0.15	6.92	0.40
NO_3-N (PPM)	20.50	10.38	39.10	6.10	52.20	8.67	17.20	7.03	11.80	3.29	3.60	3.12
NH_4-N (PPM)	65.00	5.30	76.00	7.85	47.00	4.83	44.50	4.77	63.50	8.68	99.50	11.02
PO_3-P (PPM)	31.80	2.94	32.60	1.69	37.20	2.56	46.20	3.07	17.60	1.47	23.00	3.35
August 2017 C2	A1		A2		B1		B2		OM		SSSI*	
	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM	Mean	± SEM
Soil Moisture (%)	34.72	3.51	34.48	2.69	24.33	2.16	25.74	1.04	23.35	0.48	33.91	1.67
pH	7.31	0.03	7.33	0.04	7.12	0.03	7.26	0.03	6.95	0.06	7.37	0.03
EC (dS/m)	5.85	0.41	5.81	0.40	5.14	0.78	4.53	0.36	3.71	0.21	5.33	0.31
SOC (%)	6.48	0.91	5.38	0.88	4.40	0.28	3.14	0.19	3.28	0.09	6.43	0.27
NO_3-N (PPM)	53.80	21.75	56.90	14.42	15.70	4.66	9.00	2.72	3.00	1.19	4.20	1.96
NH_4-N (PPM)	84.00	9.64	93.50	12.69	50.50	3.10	40.00	5.53	43.00	7.13	72.00	8.38
PO_3-P (PPM)	29.80	2.52	25.40	2.01	36.60	6.12	44.00	2.76	17.80	1.69	18.20	1.69

3.2.2.1 Soil moisture

In 2015 there was a significant difference in % soil moisture between plots in C1 cores (one-way ANOVA, $F(5, 24) = 13.36$, $p < 0.001$). Tukey pairwise comparisons showed that the SSSI % moisture in C1 was significantly higher than all plots, apart from A2. A2 was not significantly different to A1, but A2 C1 cores were significantly higher in moisture than B1, B2 and OM. A difference could not be detected between A1, B2 and OM, but moisture was significantly higher than B1 (Figure 3.6a). There was also a significant difference between plots in C2 cores (one-way ANOVA, $F(5, 24) = 8.11$, $p < 0.001$). The SSSI was significantly higher in % moisture in C2, but no difference was detected between the other plots (Figure 3.6b).

There was a significant difference in % soil moisture between plots in C1 cores in 2016 (one-way ANOVA, $F(5, 24) = 21.61$, $p < 0.001$). The Site A plots were not significantly different to the SSSI, but soil moisture was significantly higher than it was on the Site B plots and OM (Figure 3.6c). Soil moisture in C2 cores were significantly different (one-way ANOVA, $F(5, 24) = 13.75$, $p < 0.001$). A1 was not significantly different from A2 and the SSSI. However, soil moisture in A1 was not significantly higher than in B2. A difference could not be detected between soil moisture in B1, B2 and OM (Figure 3.6d).

In 2017 there was a significant difference in % soil moisture between plots in C1 cores (one-way ANOVA, $F(5, 24) = 20.53$, $p < 0.001$). A difference was not detected between A1, A2 and the SSSI, but both Site A plots were not significantly higher in % soil moisture than B2, which was not significantly different to that in B1. OM had the lowest % soil moisture and was different to all other plots (Figure 3.6e). There was also a significant difference between plots in August 2017 C2 cores (one-way ANOVA, $F(5, 24) = 6.38$, $p = 0.001$). There was no difference between the Site A plots and the SSSI. The Site A plots and the SSSI were significantly higher in % soil moisture than B1 and OM, but moisture levels were not significantly higher than in B2. OM was not significantly different to the Site B plots (Figure 3.6d).

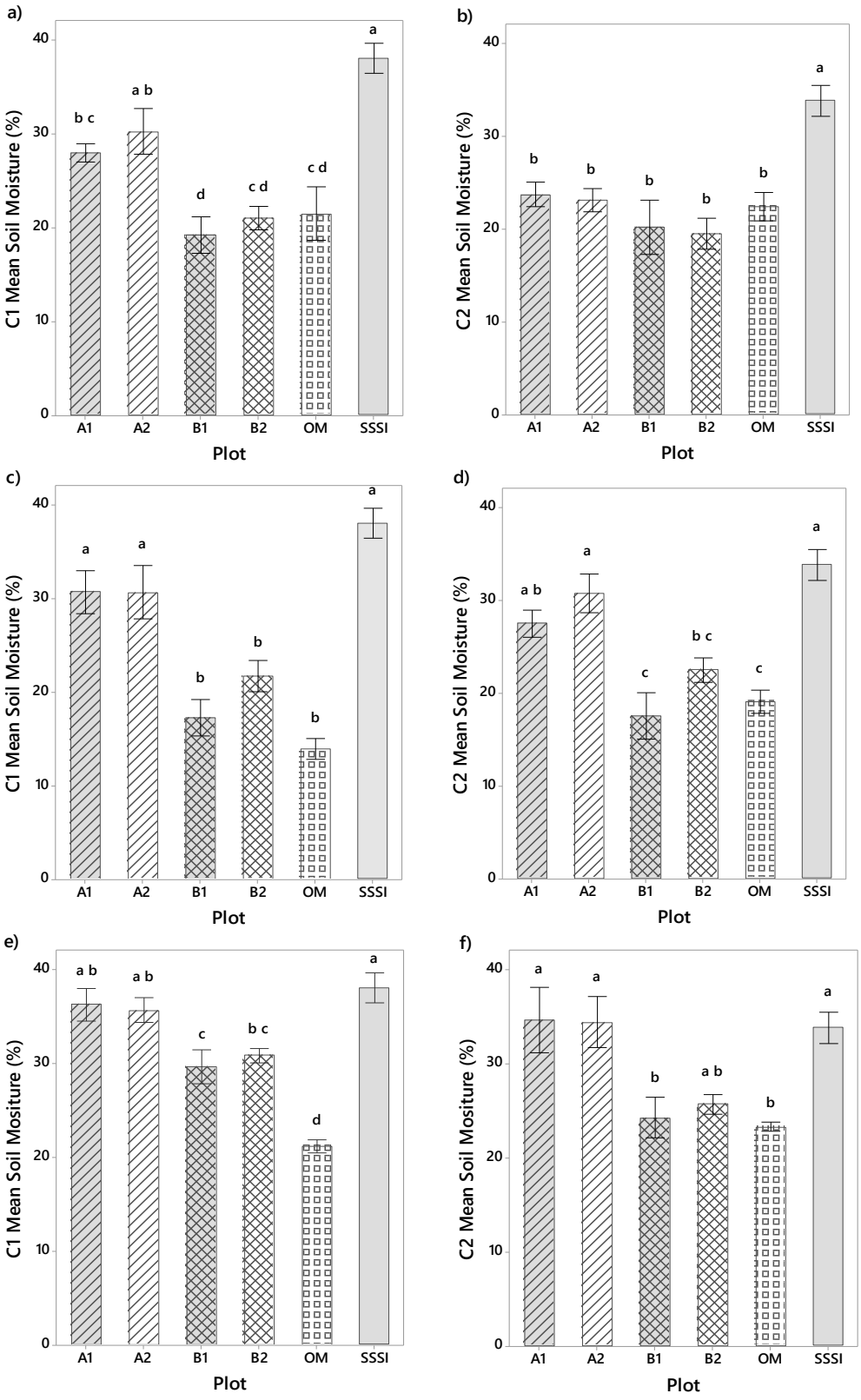


Figure 3.6 Soil moisture % in the soil of each study plot each year in a) C1 cores August 2015 (n = 5 for each plot); b) C2 cores August 2015 (n = 5); c) C1 cores August 2016 (n = 5); d) C2 cores August 2016 (n = 5); e) C1 cores August 2017 (n = 5); f) C2 cores August 2017 (n = 5). Error bars represent standard error of the mean. Means within each chart that have different letters signify significant differences ($P < 0.05$). SSSI data are from August 2016.

3.2.2.2 Soil pH

There was a significant difference in pH between plots in August 2015 (one-way ANOVA, $F(5, 24) = 9.61$, $p < 0.001$). Tukey pairwise comparisons showed that there were no differences between the pH of the SSSI and the A and B plots, but pH in OM was significantly lower (Figure 3.7a). pH was also different between plots in C2 cores (one-way ANOVA, $F(5, 24) = 4.48$, $p = 0.005$). Similarly to C1 cores, the SSSI was not different to the Site A or B plots, but it was different to OM. However, differences could not be detected between A1, B1, B2 and OM (Figure 3.7b).

In August 2016 C1 soil core samples it was found that there was a significant difference in pH between plots (one-way ANOVA, $F(5, 24) = 9.24$, $p < 0.001$). The A and B plots were similar to the SSSI, but OM was different to all plots apart from B1 (Figure 3.7c). C2 cores followed a similar pattern to C1 cores (one-way ANOVA, $F(5, 24) = 6.12$, $p = 0.001$), with differences detected between the same plots (Figure 3.7d).

pH was different between plots in August 2017 C1 cores (one-way ANOVA, $F(5, 24) = 21.33$, $p < 0.001$). B1 was different to the Site A plots and the SSSI target state. A difference was not detected between B1 and B2, but B2 was not different from Site A plots or the SSSI. pH levels in OM C1 soil cores was different to all other plots (Figure 3.7e). Differences in pH between plots were also detected in C2 cores (one-way ANOVA, $F(5, 24) = 17.57$, $p < 0.001$). The differences in pH between plots were similar to the C1 cores, but B1 could not be seperated from OM, which had significantly lower pH than all other plots (Figure 3.7f).

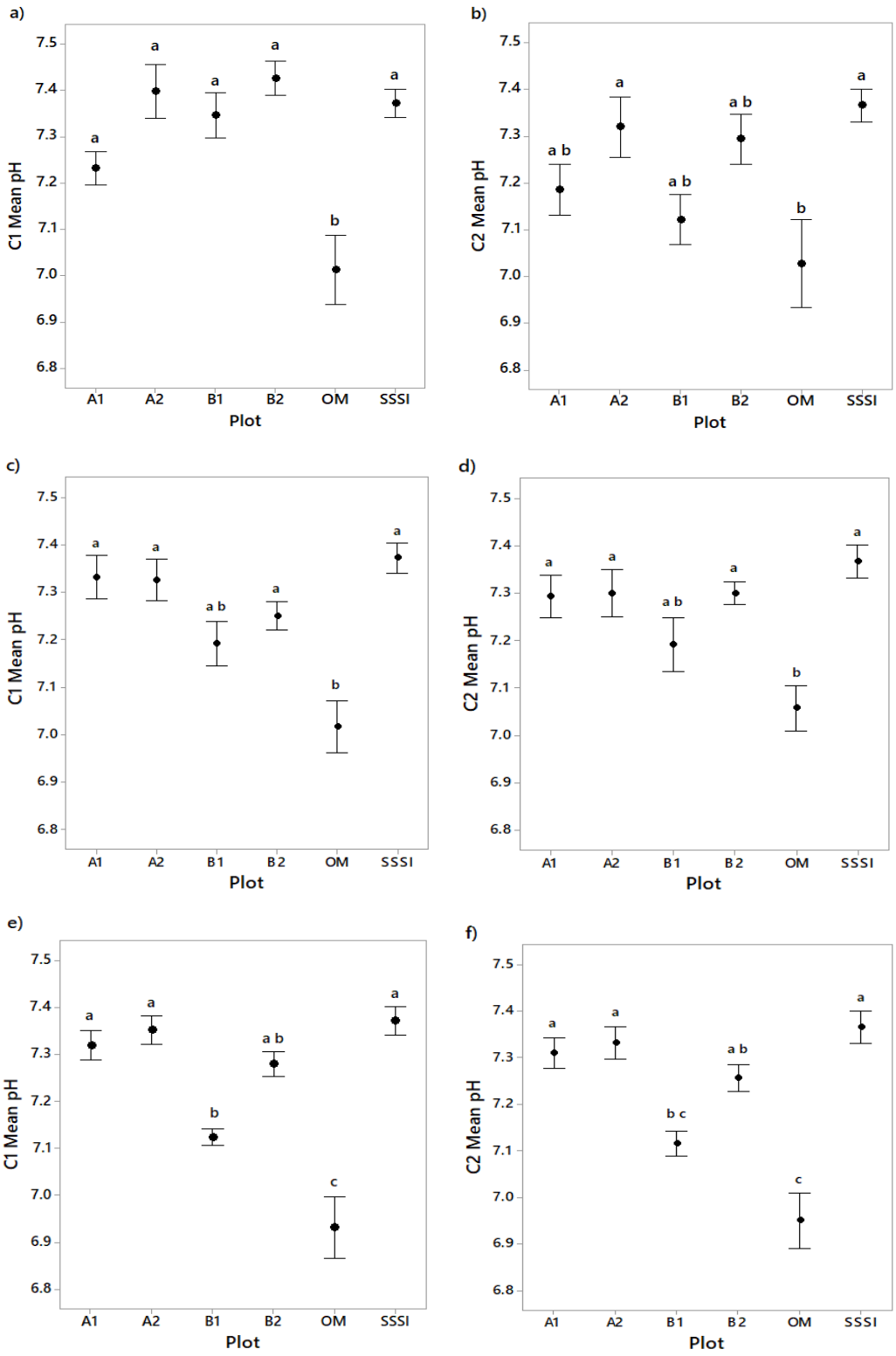


Figure 3.7 Average soil pH each year (2015-2017) and each soil depth (C1 and C2) post-breach in a) C1 cores August 2015 (n = 5 for each plot); b) C2 cores August 2015 (n = 5); c) C1 cores August 2016 (n = 5); d) C2 cores August 2016 (n = 5); e) C1 cores August 2017 (n = 5); f) C2 cores August 2017 (n = 5). Error bars represent standard error of the mean. Means within each interval plot that have different letters signify significant differences ($P < 0.05$). SSSI data are from August 2016.

3.2.2.3 Electrical conductivity

Almost one year after the breach, there was a significant difference in EC between plots in C1 cores (one-way ANOVA, $F(5, 24) = 16.04$, $p < 0.001$). Tukey pairwise comparisons showed that EC was significantly higher on the SSSI than all other sites, apart from B2. There was no detectable difference between A1, A2, B1 and B2 in August 2015. On OM, EC was significantly lower than all other plots (Figure 3.8a). A significant difference was also detected in C2 cores (one-way ANOVA, $F(5, 24) = 25.23$, $p < 0.001$). The SSSI had significantly higher EC than all other plots. There were no detectable differences between the Site A and B plots, and OM had significantly lower EC than all other plots (Figure 3.8b).

In 2016, there was a significant difference in EC between plots in C1 cores (one-way ANOVA, $F(5, 24) = 7.70$, $p < 0.001$). There was no difference between the SSSI and the Site A and B plots. OM was different to all other plots apart from A1 (Figure 3.8c). A significant difference was also detected between plots in C2 cores in August 2016 (one-way ANOVA, $F(5, 24) = 12.27$, $p < 0.001$), and Tukey post-hoc tests showed that there was no difference between the SSSI and Site A and B plots, which all had significantly higher EC than OM (Figure 3.8d).

A significant difference in EC between plots was also detected in C1 cores in August 2017 (one-way ANOVA, $F(5, 24) = 4.31$, $p = 0.006$). There were no differences between the A and B plots and the August 2016 EC on the SSSI. OM was no longer different to A1, B1, and B2 plots (Figure 3.8e). There was also a significant difference between plots in C2 cores (one-way ANOVA, $F(5, 24) = 3.37$, $p = 0.019$). There was no difference in EC between the A and B plots and the SSSI, but OM was no longer different to the SSSI, B1 and B2. Both Site A plots had higher means than the SSSI in C2 by 2017, but these were not significantly higher (Figure 3.8f).

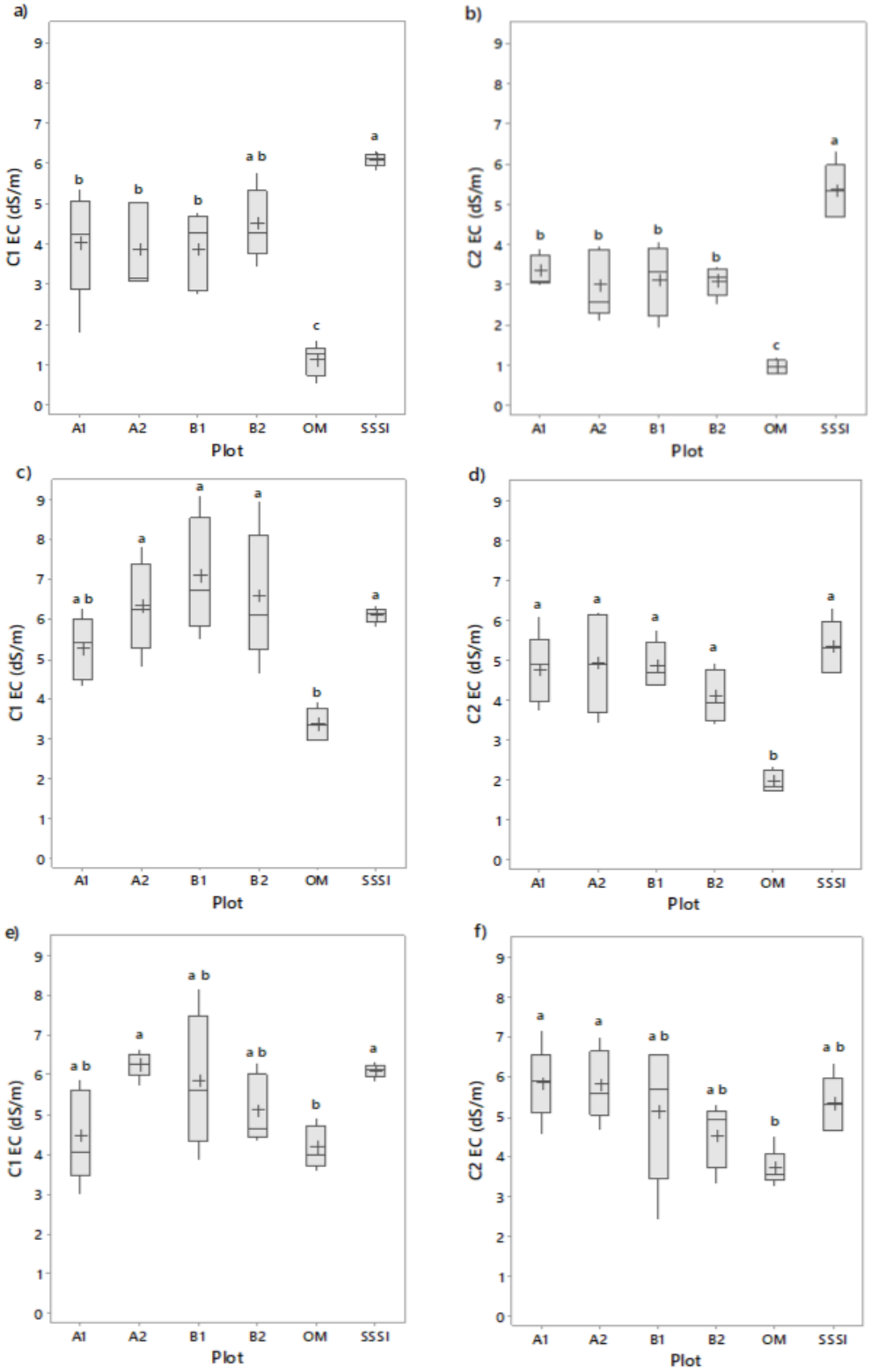


Figure 3.8 Electrical conductivity levels in the soil of each study plot each year in a) C1 cores August 2015 (n = 5 for each plot); b) C2 cores August 2015 (n = 5); c) C1 cores August 2016 (n = 5; d) C2 cores August 2016 (n = 5; e) C1 cores August 2017 (n = 5; f) C2 cores August 2017 (n = 5). Horizontal lines are medians, boxes represent the interquartile range (25-75%) and bars are minimum and maximum values. Means (+ symbol) within each boxplot that have different letters signify significant differences ($P < 0.05$). SSSI data are from August 2016. Boxplots are depicted to show a direct comparison with non-parametric pre-breach data.

3.2.2.4 Soil organic carbon (% SOC)

In 2015 there was a significant difference in % SOC between plots in C1 cores (one-way ANOVA, $F(5, 24) = 8.52$, $p < 0.001$). Tukey pairwise comparisons showed that % SOC was significantly higher on the SSSI than B1, B2 and OM, but not A1 and A2. However, there was no detectable difference between A1, A2, B1, B2 and OM in August 2015 (Figure 3.9A(i)). A significant difference was also detected in C2 cores (Kruskal-Wallis, $H(5) = 51.58$, $p = 0.008$). The SSSI had significantly higher SOC than B1, B2 and OM in these cores, but were not different to the Site A plots. As with C1 cores, there was no detectable difference between B1, B2, OM and the Site A plots (Figure 3.9B(i)).

In 2016, there was a significant difference in % SOC between plots in C1 cores (one-way ANOVA, $F(5, 24) = 42.89$, $p < 0.001$). There was a difference between the SSSI and the A and B plots, because % SOC was higher on the target state. OM was different to all other plots apart from B1 (Figure 3.9A(ii)). A significant difference was also detected between plots in C2 cores in August 2016 (Kruskal-Wallis, $H(5) = 12.65$, $p = 0.027$), and Dunn post-hoc tests showed that there was no difference between the SSSI and all other plots, apart from B1, which all had significantly lower % SOC, although no difference was detected between B1 and the other Steart Marsh and OM plots (Figure 3.9B(ii)).

A significant difference in % SOC between plots was also detected in C1 cores in August 2017 (one-way ANOVA, $F(5, 24) = 10.58$, $p < 0.001$). There were no differences between A1 and the SSSI, but all other plots were different to the target state. B2 and OM were significantly lower in % SOC than A1 and the SSSI, but these plots were not significantly lower in % SOC than A2 and B1 (Figure 3.9A(iii)). There was also a significant difference between the Site A plots and the SSSI plot in C2 cores (Kruskal-Wallis, $H(5) = 19.60$, $p = 0.001$). There was no difference in % SOC between the A plots, B1 and the SSSI. B2 and OM had significantly lower % SOC than SSSI C2 cores, but a difference could not be detected between B2, OM and A2 and B1 (Figure 3.9B(iii)).

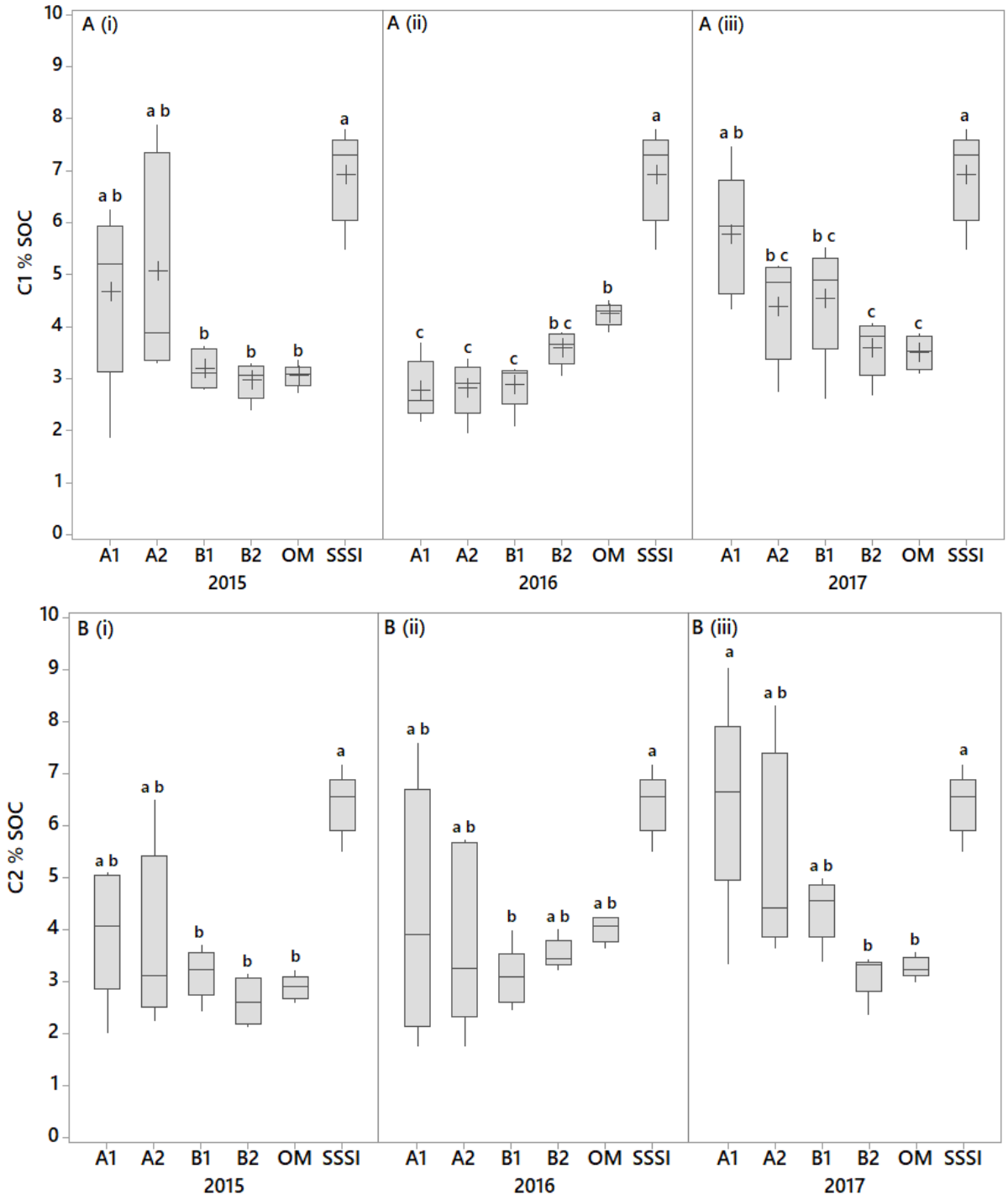


Figure 3.9 % SOC levels in the soil of each study plot each year. A (i)-(iii) = C1 cores in 2015, 2016 and 2017 (n = 5 for each plot); B (i)-(iii) = C2 cores in 2015, 2016 and 2017. A (i)-(iii) were tested by one-way ANOVA and Tukey post-hoc tests. B (i)-(iii) were tested by Kruskal-Wallis and Dunn post-hoc tests with Holm correction. Cross symbols (+) in A (i)-(iii) are means and horizontal lines are medians. Boxes represent the interquartile range (25-75%) and bars are minimum and maximum values. SSSI data are from August 2016. Means (+) that do not share letters in A (i)-(iii) are significantly different ($p < 0.05$) in each year. Medians in B (i)-(iii) that do not share letters are significantly different ($p < 0.05$) in each year.

3.2.2.5 Nitrate-nitrogen

In 2015 Nitrate-nitrogen was very variable in each plot as it had been in 2014 with a significant difference in NO_3-N between plots in C1 cores (Kruskal-Wallis, H (5) = 18.47, p = 0.002). Dunn's all-pairs test with Holm correction showed that the only detectable difference was between A1 and B2, with NO_3-N being higher in the latter. There was no difference between the Steart Marsh plots, OM or the SSSI (Figure 3.10A(ii)). A difference was found between plots in C2 cores (Kruskal-Wallis, H (5) = 11.68, p = 0.039), but post-hoc tests could not determine which plots were different.

In 2016, a difference was detected in NO_3-N between plots in C1 cores (Kruskal-Wallis, H (5) = 13.08, p = 0.023), but post-hoc tests could not determine which plot was different. The plots that were closest to being significantly different were A1 and the SSSI (p = 0.098). There was not a significant difference in NO_3-N between the plots in C2 cores in 2016 (Kruskal-Wallis, H (5) = 6.42, p = 0. 268).

A significant difference in NO_3-N between plots was detected in C1 cores in August 2017 (Kruskal-Wallis, H (5) = 16.47, p = 0.006). A2 and B1 were significantly higher than the SSSI, but not different to the other plots (Figure 3.10A(iii)). There was also a significant difference between plots in C2 cores (Kruskal-Wallis, H (5) = 19.69, p = 0.001). NO_3-N levels in A2 were significantly higher than in OM and the SSSI, but not different than the other plots. A1 was not significantly different to the SSSI, but it was different to OM (Figure 3.10B(iii)).

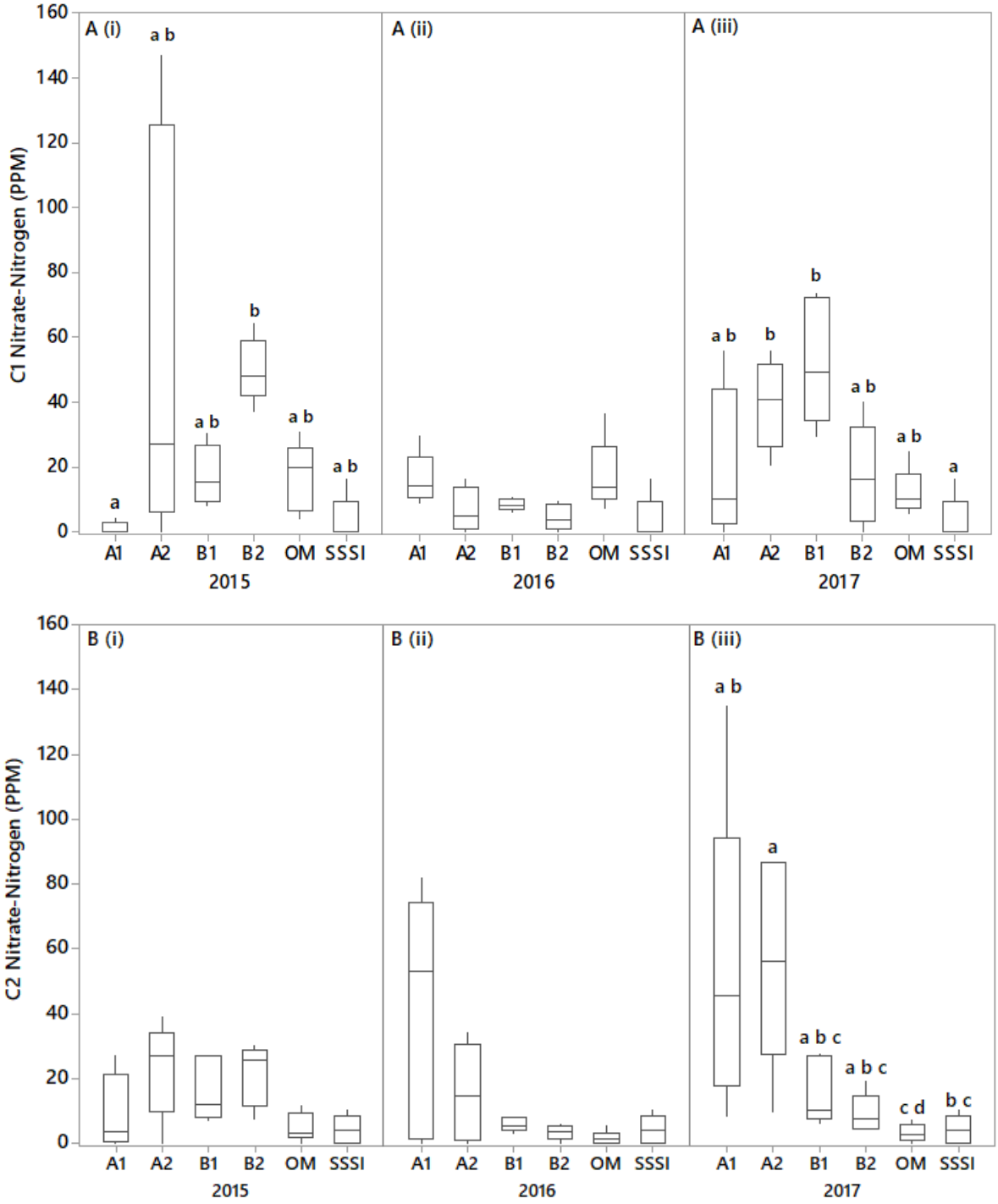


Figure 3.10 Nitrate-nitrogen levels in the soil of each study plot each year. A (i)-(iii) = C1 cores in 2015, 2016 and 2017 (n = 5 for each plot); B (i)-(iii) = C2 cores in 2015, 2016 and 2017 (n = 5 for each plot). Horizontal lines are medians, boxes represent the interquartile range (25-75%) and bars are minimum and maximum values. SSSI data are from August 2016. Medians that do not share letters in A (i), A(iii) and B(iii) are significantly different ($p < 0.05$) in each year. Plots in each year were tested with Kruskal-Wallis and Dunn post-hoc tests with Holm correction where appropriate. There was not a significant difference between plots in boxplots that do not have letters.

3.2.2.6 Ammonium-nitrogen

In 2015 there was not a significant difference in NH_4-N between plots in C1 cores (one-way ANOVA, $F(5, 24) = 1.90$, $p = 0.131$) or C2 cores (one-way ANOVA, $F(5, 24) = 1.11$, $p = 0.383$).

In 2016, a difference was detected in NH_4-N between plots in C1 cores (one-way ANOVA, $F(5, 24) = 7.74$, $p < 0.001$). All plots were significantly lower in NH_4-N than the SSSI target state (Figure 3.11A(ii)). There was also a significant difference in NH_4-N between the plots in C2 cores in 2016 (one-way ANOVA, $F(5, 24) = 3.18$, $p = 0.024$). B1 has significantly lower NH_4-N levels than A1 and the SSSI. There were no differences between the other plots (Figure 3.11B(ii))

A significant difference in NH_4-N between plots was detected in C1 cores in August 2017 (one-way ANOVA, $F(5, 24) = 7.40$, $p < 0.001$). All plots apart from A2 were significantly lower in NH_4-N than the SSSI, but A2 was not significantly different to the other plots (Figure 3.11A(iii)). There was also a significant difference in NH_4-N between the plots in C2 cores (one-way ANOVA, $F(5, 24) = 7.32$, $p < 0.001$). NH_4-N levels in A1 and A2 were significantly higher than in B1, B2 and OM, but there was no difference between the SSSI and all other plots (Figure 3.11B(iii)).

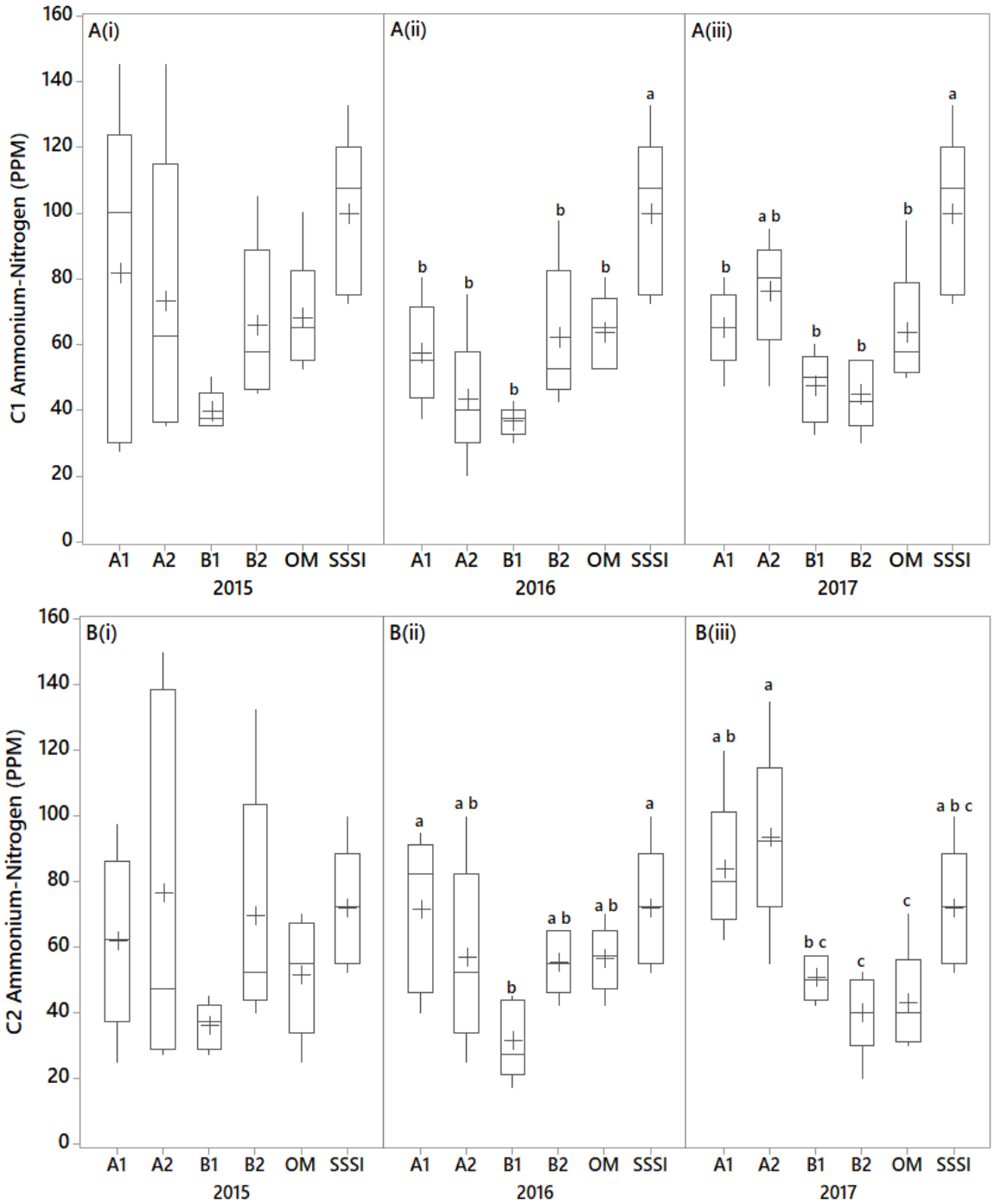


Figure 3.11 Ammonium-nitrogen levels in the soil of each study plot each year. A (i)-(iii) = C1 cores in 2015, 2016 and 2017 (n = 5 for each plot); B (i)-(iii) = C2 cores in 2015, 2016 and 2017 (n = 5 for each plot). Horizontal lines are medians, boxes represent the interquartile range (25-75%) and bars are minimum and maximum values. Means (+ symbol) within each year that have different letters signify significant differences ($p < 0.05$). SSSI data are from August 2016. There was not a significant difference between plots in boxplots that do not have letters.

3.2.2.7 Phosphate-phosphorus

In 2015 there was a significant difference in PO_3-P between plots in C1 cores (one-way ANOVA, $F(5, 24) = 6.51$, $p = 0.001$). B1 and B2 were not different to the SSSI. Average PO_3-P was similar in these plots and was higher than in the Site A plots and OM. However, only B2 had significantly higher PO_3-P than these plots (Figure 3.12A(i)). There was also a significant difference in PO_3-P between plots in C2 cores (one-way ANOVA, $F(5, 24) = 6.14$, $p = 0.001$). Like C1 cores, B2 had significantly higher levels of PO_3-P than A1, A2 and OM, but it did not have significantly higher levels than B1 and the SSSI (Figure 3.12B(i))

In 2016, a difference was detected in PO_3-P between plots in C1 cores (one-way ANOVA, $F(5, 24) = 3.58$, $p = 0.015$). There was no difference between the Site A plots, the Site B plots and the SSSI. OM had the lowest levels of PO_3-P and this was significantly lower than levels in A1 and B2. However, there was no detectable difference between OM and A2, B1 and the SSSI (Figure 3.12A(ii)). There was a significant difference in PO_3-P between the plots in C2 cores in 2016 (one-way ANOVA, $F(5, 24) = 6.40$, $p = 0.001$). B2 had significantly higher PO_3-P levels than all other plots apart from B1. There was no difference between the other plots (Figure 3.12B(ii))

A significant difference in PO_3-P between plots was detected in C1 cores in August 2017 (one-way ANOVA, $F(5, 24) = 15.14$, $p < 0.001$). B1 and B2 had the highest levels of PO_3-P in C1 cores and these were not significantly different to one another. However, B1 did not have significantly higher PO_3-P than both Site A plots. The SSSI had lower PO_3-P measurements than both Site B plots but was not different than the Site A plots. OM had significantly lower levels than all plots apart from the SSSI (Figure 3.12A(iii)). There was also a significant difference in PO_3-P between the plots in C2 cores (one-way ANOVA Welsh's Test, $F(5) = 14.19$, $p < 0.001$; equal variances not assumed). Like C1 cores, B1 and B2 had the highest levels of PO_3-P in C2 cores and these were not significantly different to one another. B2 had higher levels than all other plots, but B1 levels were not significantly higher than the other plots. There was no detectable difference between the Site A plots, OM and the SSSI (Figure 3.12B(iii)).

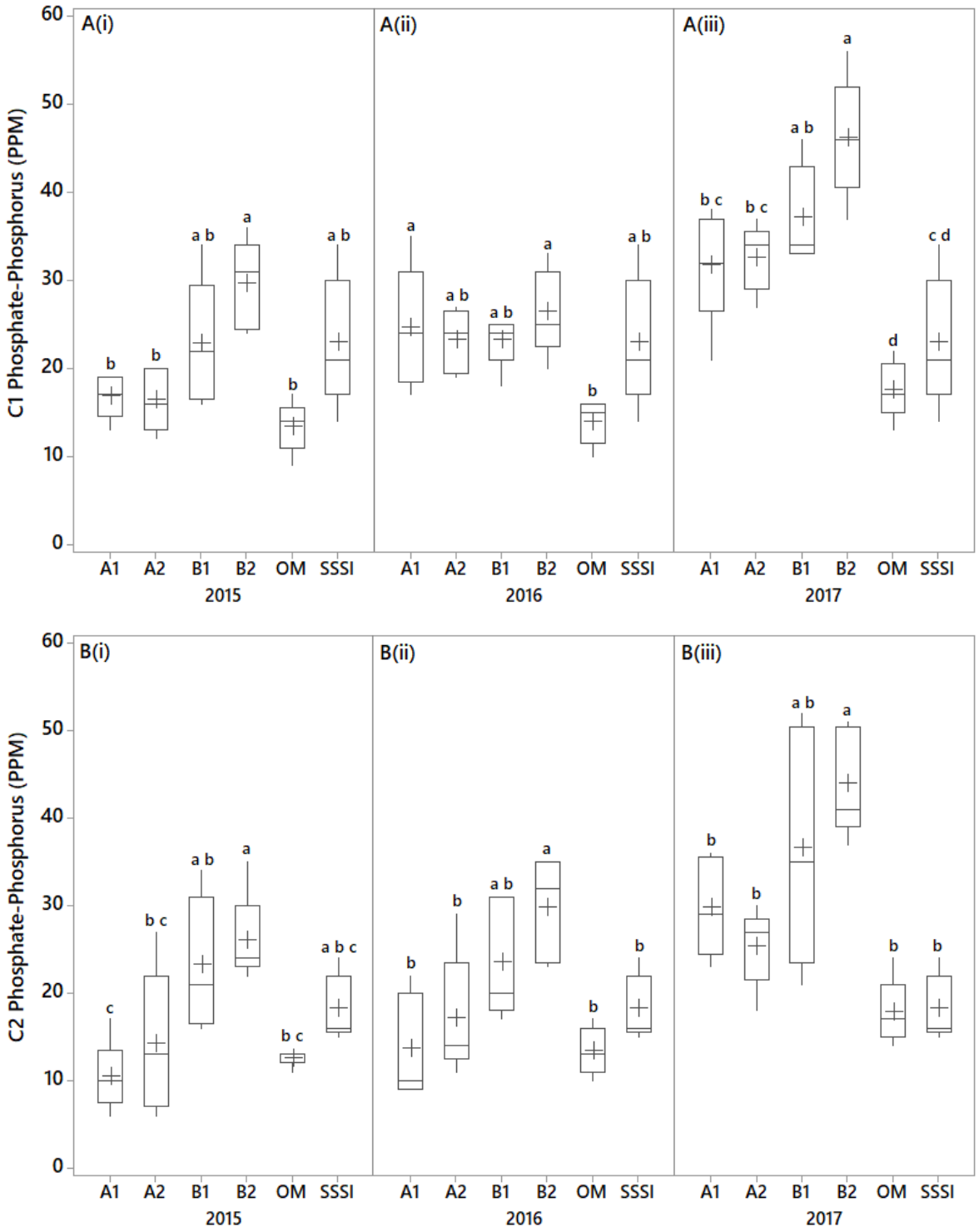


Figure 3.12 Phosphate-phosphorous levels in the soil of each study plot each year. A (i)-(iii) = C1 cores in 2015, 2016 and 2017 (n = 5 for each plot); B (i)-(iii) = C2 cores in 2015, 2016 and 2017 (n = 5 for each plot). Horizontal lines are medians, boxes represent the interquartile range (25-75%) and bars are minimum and maximum values. Means (+ symbol) within each year that have different letters signify significant differences ($p < 0.05$). SSSI data are from August 2016. One-way ANOVA with Tukey post-hoc tests were used for each year apart from B(iii) data that was tested by one-way ANOVA Welsh's Test with a Games-Howell post-hoc test.

3.2.3 Soil characteristics – differences between years

3.2.3.1 Site A

On the Site A quadrats soil moisture and EC were lower in 2014 as expected. Both variables increased post-breach, and surpassed the measurements found in SSSI cores by 2016 in most quadrats. The five other soil variables varied more between quadrats over the years, but there were similarities between the quadrats and the SSSI target state. There was not a clear pattern to pH levels throughout the study period and SOC varied in comparison to the SSSI, with some quadrats having similar % SOC as the target state by 2017 and others having lower levels. Nitrate-nitrogen levels were higher in 2014 than subsequent years in most C1 cores, but this was not the case in 90% of the C2 quadrats. Like nitrogen-nitrate levels, ammonium-nitrogen varied between the quadrats, but there were similarities with the SSSI target state in some quadrats post-breach. In most of the quadrats, phosphate-phosphorus was higher in 2017 than it had been in previous years (Figure 3.13).

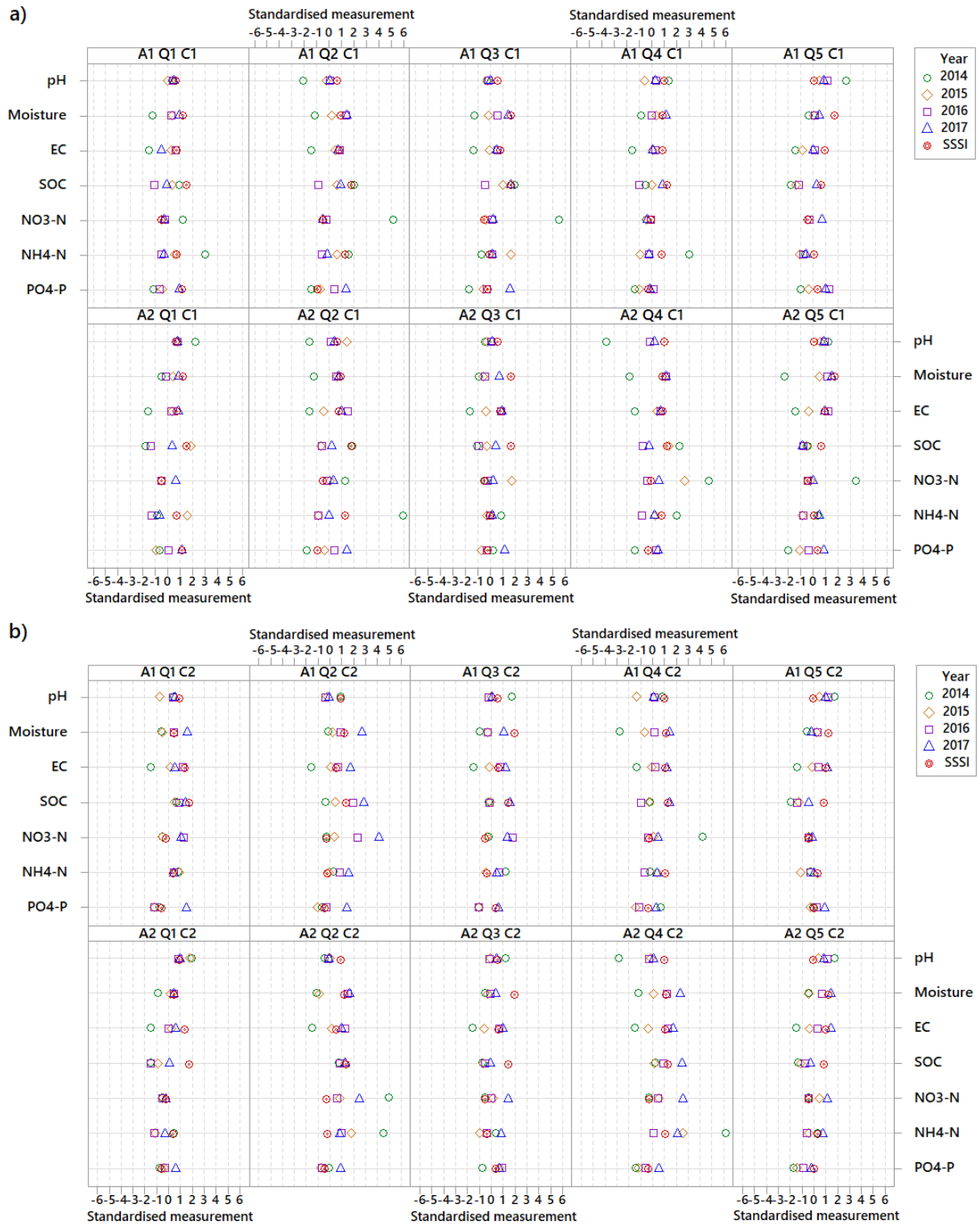


Figure 3.13 Individual value plots of soil variables by year (2014-2017) in a) A1 and A2 C1 cores, and b) A1 and A2 C2 cores. Individual values are based on standardised measurements because soil variables had different measurement scales. SSSI is the target state and based on C1 Q1-Q5 and C2 Q1-Q5 (recorded in August 2016).

3.2.3.1.1 Plot A1

There were no significant differences detected between years for pH in C1 cores, and SOC, NO_3-N and NH_4-N in both C1 and C2 cores. There were significant differences in moisture between years in C1 ($p < 0.001$) and C2 ($p = 0.003$), with 2017 having the highest % moisture in cores (Table 3.3). Effects were also detected in EC C1 ($p < 0.001$) and C2 ($p < 0.001$). Salt levels were the highest in 2017 in both cores, but differences were only detected between 2014 and the three subsequent years in C1. There were no significant differences between 2015, 2016 and 2017 in C1, but 2016 and 2017 did differ from 2014 and 2015 in C2, and a difference was detected between 2014 and 2015 in C2 (Table 3.3). PO_4-P levels were significantly different between years in C1 ($p < 0.001$) and C2 ($p < 0.001$). The highest phosphate levels were found in 2017 cores. These were not significantly different to 2016 levels in C1 cores, but they did differ from 2014 and 2015 levels. However, 2017 levels were significantly different to all other years in C2 because they were substantially higher (Table 3.3).

3.2.3.1.2 Plot A2

There were no significant differences detected between years for pH, SOC, NO_3-N and NH_4-N in both C1 and C2 cores. There were significant differences in moisture between years in C1 ($p < 0.001$) and C2 ($p < 0.001$), with 2017 having the highest % moisture in cores (Table 3.4). Effects were also detected in EC C1 ($p < 0.001$) and C2 ($p < 0.001$). Salt levels were the highest in 2016 and 2017 in both cores. There were no differences detected between these years in C1 or C2, but EC did differ between 2014 and 2015, and 2016 and 2017 levels did differ from 2015 (Table 3.4). PO_4-P levels were significantly different between years in C1 ($p < 0.001$) and C2 ($p < 0.024$). The highest phosphate levels were found in 2017 cores. These were not significantly different to 2016 levels in C1 cores, but they did differ from 2014 and 2015 levels. In C2 cores, 2017 levels were significantly different to 2014 levels but not from 2014 and 2015 levels (Table 3.4).

Table 3.3 Differences in soil variables between years (2014-2017) on A1 in C1 (top 10cm) and C2 (10-20cm), n = 5 for each variable and each core depth. * = one-way ANOVA using Tukey post hoc-tests if significant differences were detected; ** = Kruskal-Wallis test; ^{ns} = not significant. Average = Mean ± SEM for ANOVA or Median (Mdn) for Kruskal-Wallis test. In post-hoc grouping column, means or medians in each core depth that do not share letters are significantly different (< 0.05), ns = not significant. Non-standardised data were used for tests because the variables were tested separately by year.

A1 - soil variable versus year				Post-hoc analysis				
Soil variable	df	Test statistic	P-value	Year	Average		Grouping	
					C1	C2	C1	C2
Moisture (%)				2014	17.44 ± 1.42	17.94 ± 3.43	c	B
C1 *	3, 16	F = 22.71	< 0.001	2015	28.04 ± 0.97	23.75 ± 1.35	b	B
C2 *	3, 16	F = 7.05	0.003	2016	30.78 ± 2.29	27.57 ± 1.48	a b	a b
				2017	36.30 ± 1.68	34.72 ± 3.51	a	A
pH				2014	n/a	7.45 ± 0.04	n/a	A
C1 **	3	H = 2.82	0.420 ^{ns}	2015	n/a	7.19 ± 0.05	n/a	B
C2 *	3, 16	F = 6.03	0.006	2016	n/a	7.29 ± 0.05	n/a	a b
				2017	n/a	7.31 ± 0.03	n/a	a b
EC (dS/m)				2014	0.48 ± 0.10	0.31 ± 0.05	b	C
C1 *	3, 16	F = 23.27	< 0.001	2015	4.03 ± 0.60	3.35 ± 0.17	a	B
C2 *	3, 16	F = 64.10	< 0.001	2016	5.27 ± 0.36	4.78 ± 0.40	a	A
				2017	4.46 ± 0.52	5.85 ± 0.41	a	A
SOC (%)				2014	n/a	n/a	n/a	n/a
C1 *	3, 16	F = 2.47	0.099 ^{ns}	2015	n/a	n/a	n/a	n/a
C2 *	3, 16	F = 2.46	0.101 ^{ns}	2016	n/a	n/a	n/a	n/a
				2017	n/a	n/a	n/a	n/a
NO_3-N (ppm)				2014	n/a	n/a	n/a	n/a
C1 **	3	H = 5.92	0.115 ^{ns}	2015	n/a	n/a	n/a	n/a
C2 **	3	H = 4.12	0.249 ^{ns}	2016	n/a	n/a	n/a	n/a
				2017	n/a	n/a	n/a	n/a
NH_4-N (ppm)				2014	n/a	n/a	n/a	n/a
C1 *	3, 16	F = 1.38	0.316 ^{ns}	2015	n/a	n/a	n/a	n/a
C2 *	3, 16	F = 0.73	0.547 ^{ns}	2016	n/a	n/a	n/a	n/a
				2017	n/a	n/a	n/a	n/a
PO_4-P (ppm)				2014	9.80 ± 1.24	16.60 ± 2.98	c	B
C1 *	3, 16	F = 17.23	< 0.001	2015	16.80 ± 1.11	10.40 ± 1.81	b c	B
C2 *	3, 16	F = 11.28	< 0.001	2016	24.60 ± 3.11	13.60 ± 2.69	a b	B
				2017	31.80 ± 2.94	29.80 ± 2.52	a	A

Table 3.4 Differences in soil variables between years (2014-2017) on A2 in C1 (top 10cm) and C2 (10-20cm), n = 5 for each variable and each core depth. * = one-way ANOVA using Tukey post hoc-tests if significant differences were detected; ** = Kruskal-Wallis test; ^{ns} = not significant. Average = Mean ± SEM for ANOVA or Median (Mdn) for Kruskal-Wallis test. In post-hoc grouping column, means or medians in each core depth that do not share letters are significantly different (< 0.05), ns = not significant. Non-standardised data were used for tests because the variables were tested separately by year.

A2 - soil variable versus year				Post-hoc analysis				
Soil variable	df	Test statistic	P-value	Year	Average		Grouping	
					C1	C2	C1	C2
Moisture (%)				2014	14.01 ± 2.98	19.91 ± 1.05	b	b
C1 *	3, 16	F = 14.39	< 0.001	2015	30.29 ± 2.48	23.17 ± 1.29	a	b
C2 *	3, 16	F = 13.71	< 0.001	2016	30.69 ± 2.86	30.82 ± 2.05	a	a
				2017	35.73 ± 1.26	34.48 ± 2.69	a	a
pH				2014	n/a	n/a	n/a	n/a
C1 **	3	H = 1.32	0.725 ^{ns}	2015	n/a	n/a	n/a	n/a
C2 *	3, 16	F = 0.02	0.995 ^{ns}	2016	n/a	n/a	n/a	n/a
				2017	n/a	n/a	n/a	n/a
EC (dS/m)				2014	0.39 ± 0.09	0.22 ± 0.05	c	c
C1 *	3, 16	F = 59.37	< 0.001	2015	3.87 ± 0.48	2.99 ± 0.37	b	b
C2 *	3, 16	F = 40.47	< 0.001	2016	6.32 ± 0.52	4.91 ± 0.55	a	a
				2017	6.26 ± 0.15	5.81 ± 0.40	a	a
SOC (%)				2014	n/a	n/a	n/a	n/a
C1 *	3, 16	F = 1.21	0.338 ^{ns}	2015	n/a	n/a	n/a	n/a
C2 *	3, 16	F = 1.28	0.316 ^{ns}	2016	n/a	n/a	n/a	n/a
				2017	n/a	n/a	n/a	n/a
NO_3-N (ppm)				2014	n/a	n/a	n/a	n/a
C1 **	3	H = 4.79	0.188 ^{ns}	2015	n/a	n/a	n/a	n/a
C2 *	3	H = 5.88	0.118 ^{ns}	2016	n/a	n/a	n/a	n/a
				2017	n/a	n/a	n/a	n/a
NH_4-N (ppm)				2014	n/a	n/a	n/a	n/a
C1 **	3	H = 7.30	0.063 ^{ns}	2015	n/a	n/a	n/a	n/a
C2 *	3, 16	F = 2.06	0.146 ^{ns}	2016	n/a	n/a	n/a	n/a
				2017	n/a	n/a	n/a	n/a
PO_4-P (ppm)				2014	12.00 ± 4.00	11.00 ± 2.98	c	b
C1 *	3, 16	F = 13.44	< 0.001	2015	16.40 ± 1.60	14.20 ± 3.73	b c	a b
C2 *	3, 16	F = 4.14	0.024	2016	23.20 ± 1.59	17.20 ± 3.15	a b	a b
				2017	32.60 ± 1.69	25.40 ± 2.01	a	a

3.2.3.2 Site B

In most of the quadrats on Site B, soil variables were not like the SSSI target state by 2017. Moisture was lower than the SSSI in C1 and C2 cores in every quadrat, although in all quadrats, moisture was higher in 2017 than it had been in previous years. As expected, EC was lowest in 2014, but it increased post-breach. In many quadrats, EC was lower in 2017 than the SSSI, but it did reach higher levels in some quadrats in both C1 and C2 cores. Like the Site A cores, pH was generally variable in each quadrat, but it was mainly alkaline (between 7 and 8) throughout the study. SOC was lower than it was in the SSSI in all C1 and C2 quadrats. Nitrate-nitrogen was higher than in the SSSI plot in most of the quadrats, but ammonium-nitrogen was the opposite, although there were similarities between measurements in quadrats. Ammonium-nitrogen tended to be lower in 2017 than it had been in previous years. Phosphate-phosphorus was higher in 2017 C1 and C2 cores than it had been in all previous years, and these levels were also higher than the SSSI (Figure 3.14).

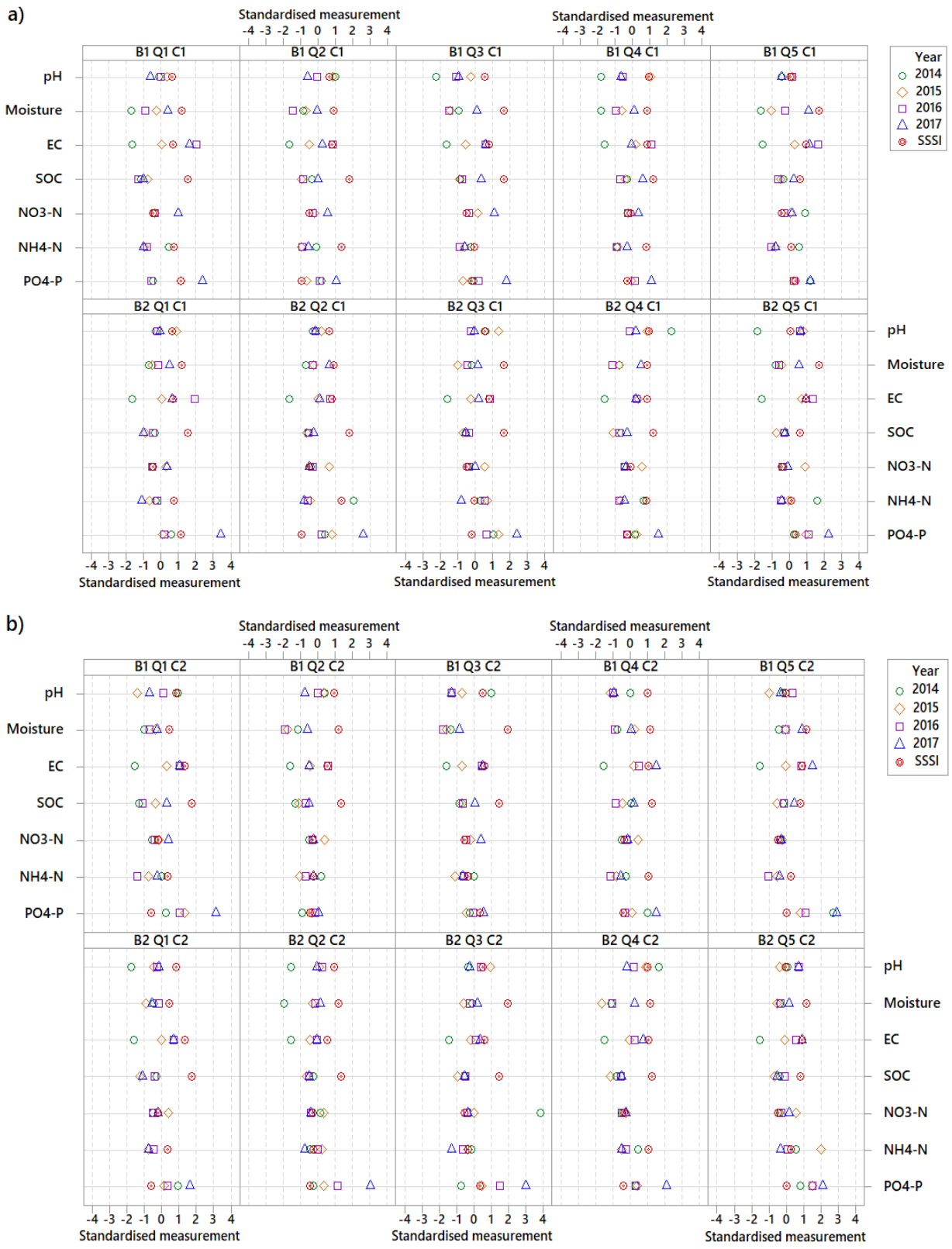


Figure 3.14 Individual value plots of soil variables by year (2014-2017) in a) B1 and B2 C1 cores, and b) B1 and B2 C2 cores. Individual values are based on standardised measurements because soil variables had different measurement scales.

3.2.3.2.1 Plot B1

There were no significant differences between years for pH in C1 cores, % moisture in C2 cores, NH_4-N levels in C1 cores, and PO_4-P levels in C2 cores. There were differences in pH between years in C2 (0.017), but there was not a clear pattern because the pH dropped in 2015, rose in 2016 and dropped again to 2015 levels in 2017 (Table 3.5). C1 % moisture was significantly different between years ($p < 0.001$), with moisture levels being considerably higher in 2017 than in the previous three years. EC was significantly different between years in C1 ($p < 0.001$) and C2 ($p < 0.001$) cores. It was relatively high in 2016 and 2017 cores, but EC was at its highest in 2016 in C1 and dropped in 2017. EC in both cores was significantly different post-breach, with levels being relatively low pre-breach. % SOC was significantly different between years in C1 ($p = 0.011$) and C2 ($p = 0.017$). It was at its highest levels in 2017, and significantly different to 2015 and 2016 in C1, but SOC in 2017 was not significantly different from 2014 pre-breach levels in C1. % SOC was significantly higher in 2017 when compared to 2014 in C2 (Table 3.5). Nitrate levels were variable across quadrats, but there was a significant difference between years in C1 ($p = 0.016$) and C2 ($p = 0.009$). 2017 levels were significantly different than the 2014 starting state in both cores, but this change appeared to be gradual because 2015 and 2016 levels were not different from 2014 in C1. In C2 nitrate levels were significantly higher than 2014 levels in 2015, but a difference was not detected in 2016, although there was variation in data. NH_4-N was significantly different between years in C2 cores ($P < 0.001$). Levels were highest before the breach and dropped significantly in 2015. Levels did not change significantly in 2016, but they were higher in 2017. In C1 core, PO_4-P was significantly different between years ($p = 0.003$). Levels were similar in 2014, 2015 and 2016, but they were significantly higher in 2017 (Table 3.5).

3.2.3.2.2 Plot B2

There were no significant differences between years for pH, % moisture in C2 cores, % SOC in C1 cores, NO_3-N levels in C2 cores, and NH_4-N levels in C2 cores. Moisture in C1 cores was significantly different between years ($p < 0.001$). Levels were not significantly different in the first two years after the breach, but they were significantly higher in 2017. EC was significantly different between years in C1 ($p < 0.001$) and C2 ($p < 0.001$). This soil characteristic was higher in the years post-breach, but differences were not as substantial between 2015, 2016 and 2017. However, in C2 cores, EC in 2017 was significantly higher than this characteristic was in 2015. SOC was significantly different between years in C2 cores. % SOC dropped in 2015 from initial levels in 2014, but data were like pre-breach

levels again in 2016 and 2017. Although NO_3-N was significantly different between years in C1, there was not a clear pattern due to fluctuations. Levels were much higher in 2015 than they were pre-breach, but they dropped substantially again in 2016, before rising again in 2017. NH_4-N levels changed significantly between years in C1 ($p = 0.022$). This variable was high in 2014, and dropped in 2015 and 2016, although this was not a significant change. However, 2017 levels were significantly lower than pre-breach levels. PO_4-P was significantly different between years in C1 (< 0.001) and C2 (< 0.001). There was not a significant change in the first two years after the breach, but 2017 levels were significantly higher.

Table 3.5 Differences in soil variables between years (2014-2017) on B1 in C1 (top 10cm) and C2 (10-20cm), n = 5 for each variable and each core depth. * = one-way ANOVA using Tukey post hoc-tests if significant differences were detected; ** = Kruskal-Wallis test using Dunn post-hoc test with Holm correction; ^{ns} = not significant. Average = Mean ± SEM for ANOVA or Median (Mdn) for Kruskal-Wallis test. In post-hoc grouping column, means or medians in each core depth that do not share letters are significantly different (< 0.05), ns = not significant. Non-standardised data were used for tests because the variables were tested separately by year.

B1 - soil variable versus year				Post-hoc analysis				
Soil variable	df	Test statistic	P-value	Year	Average		Grouping	
					C1	C2	C1	C2
Moisture (%)				2014	13.84 ± 1.93	n/a	b	n/a
C1 *	3, 16	F = 12.60	< 0.001	2015	19.24 ± 1.95	n/a	b	n/a
C2 *	3, 16	F = 1.72	0.203 ^{ns}	2016	17.37 ± 1.95	n/a	b	n/a
				2017	29.68 ± 1.84	n/a	a	n/a
pH				2014	n/a	7.33 ± 0.04	n/a	a
C1 *	3, 16	F = 2.36	0.110 ^{ns}	2015	n/a	7.12 ± 0.05	n/a	b
C2 *	3, 16	F = 4.58	0.017	2016	n/a	7.19 ± 0.06	n/a	a b
				2017	n/a	7.12 ± 0.03	n/a	b
EC (dS/m)				2014	0.16 ± 0.03	0.14 ± 0.02	c	c
C1 *	3, 16	F= 31.24	< 0.001	2015	3.86 ± 0.43	3.13 ± 0.39	b	b
C2 *	3, 16	F = 25.60	< 0.001	2016	7.10 ± 0.65	4.87 ± 0.26	a	a b
				2017	5.85 ± 0.75	5.14 ± 0.78	a b	a
SOC (%)				2014	3.37 ± 0.29	3.02 ± 0.44	a b	b
C1 *	3, 16	F = 5.15	0.011	2015	3.18 ± 0.17	3.17 ± 0.21	b	a b
C2 *	3, 16	F = 4.58	0.017	2016	2.89 ± 0.20	3.07 ± 0.26	b	b
				2017	4.54 ± 0.50	4.40 ± 0.28	a	a
NO_3-N (ppm)				2014	Mdn = 7.50	1.00 ± 0.69	a	b
C1 **	3	H = 10.37	0.016	2015	Mdn = 15.00	16.40 ± 4.40	a b	a
C2 *	3, 16	F = 5.45	0.009	2016	Mdn = 8.00	5.60 ± 0.94	a	a b
				2017	Mdn = 49.00	15.70 ± 4.66	b	a
NH_4-N (ppm)				2014	n/a	66.00 ± 3.22	n/a	a
C1 *	3, 16	F = 3.78	0.057 ^{ns}	2015	n/a	36.00 ± 3.22	n/a	b c
C2 *	3, 16	F = 16.72	< 0.001	2016	n/a	31.50 ± 5.28	n/a	c
				2017	n/a	50.50 ± 3.10	n/a	b
PO_4-P (ppm)				2014	24.00 ± 2.70	n/a	b	n/a
C1 *	3, 16	F = 7.40	0.003	2015	22.80 ± 3.25	n/a	b	n/a
C2 *	3, 16	F = 1.67	0.214 ^{ns}	2016	23.20 ± 1.32	n/a	b	n/a
				2017	37.20 ± 2.56	n/a	a	n/a

Table 3.6 Differences in soil variables between years (2014-2017) on B2 in C1 (top 10cm) and C2 (10-20cm), n = 5 for each variable and each core depth. * = one-way ANOVA using Tukey post hoc-tests if significant differences were detected; ** = Kruskal-Wallis test using Dunn post-hoc test with Holm correction if significant differences were detected; ^{ns} = not significant. Average = Mean ± SEM for ANOVA or Median (Mdn) for Kruskal-Wallis test. In post-hoc grouping column, means or medians in each core depth that do not share letters are significantly different (< 0.05), ns = not significant. Non-standardised data were used for tests because the variables were tested separately by year.

B2 - soil variable versus year				Post-hoc analysis				
Soil variable	df	Test statistic	P-value	Year	Average		Grouping	
					C1	C2	C1	C2
Moisture (%)				2014	20.83 ± 1.04	n/a	b	n/a
C1 *	3, 16	F = 15.83	< 0.001	2015	21.10 ± 1.26	n/a	b	n/a
C2 *	3, 16	F = 3.12	0.056 ^{ns}	2016	21.72 ± 1.64	n/a	b	n/a
				2017	30.91 ± 0.78	n/a	a	n/a
pH				2014	n/a	n/a	n/a	n/a
C1**	3	H = 7.48	0.058 ^{ns}	2015	n/a	n/a	n/a	n/a
C2 *	3, 16	F = 0.67	0.582 ^{ns}	2016	n/a	n/a	n/a	n/a
				2017	n/a	n/a	n/a	n/a
EC (dS/m)				2014	0.16 ± 0.02	0.21 ± 0.05	c	C
C1 *	3, 16	F = 36.74	< 0.001	2015	4.50 ± 0.39	3.10 ± 0.16	b	B
C2 *	3, 16	F = 62.41	< 0.001	2016	6.57 ± 0.73	4.09 ± 0.29	a	a b
				2017	5.11 ± 0.39	4.53 ± 0.36	a b	A
SOC (%)				2014	n/a	3.44 ± 0.19	n/a	A
C1 **	3	H = 7.29	0.063 ^{ns}	2015	n/a	2.63 ± 0.20	n/a	B
C2 *	3, 16	F = 5.14	0.011	2016	n/a	3.53 ± 0.13	n/a	A
				2017	n/a	3.14 ± 0.19	n/a	a b
NO_3-N (ppm)				2014	Mdn = 3.50	n/a	a	n/a
C1 **	3	H = 12.44	0.006	2015	Mdn = 48.00	n/a	b	n/a
C2 **	3	H = 7.48	0.058 ^{ns}	2016	Mdn = 3.50	n/a	a	n/a
				2017	Mdn = 16.00	n/a	a b	n/a
NH_4-N (ppm)				2014	Mdn = 102.50	n/a	a	n/a
C1 **	3	H = 9.65	0.022	2015	Mdn = 57.50	n/a	ab	n/a
C2 *	3, 16	F = 2.16	0.132 ^{ns}	2016	Mdn = 52.50	n/a	ab	n/a
				2017	Mdn = 42.50	n/a	b	n/a
PO_4-P (ppm)				2014	27.60 ± 1.60	22.40 ± 3.14	b	B
C1 *	3, 16	F = 15.72	< 0.001	2015	29.60 ± 2.25	26.00 ± 2.30	b	B
C2 *	3, 16	F = 12.06	< 0.001	2016	26.40 ± 2.18	29.80 ± 2.63	b	B
				2017	46.20 ± 3.07	44.00 ± 2.76	a	A

3.2.3.3 Plot OM

On OM, soil variable measurments througout the study were lower than those found on the SSSI in most of the quadrats, but this was not true of the macronutrients in all quadrats (Figure 3.15 a and b)

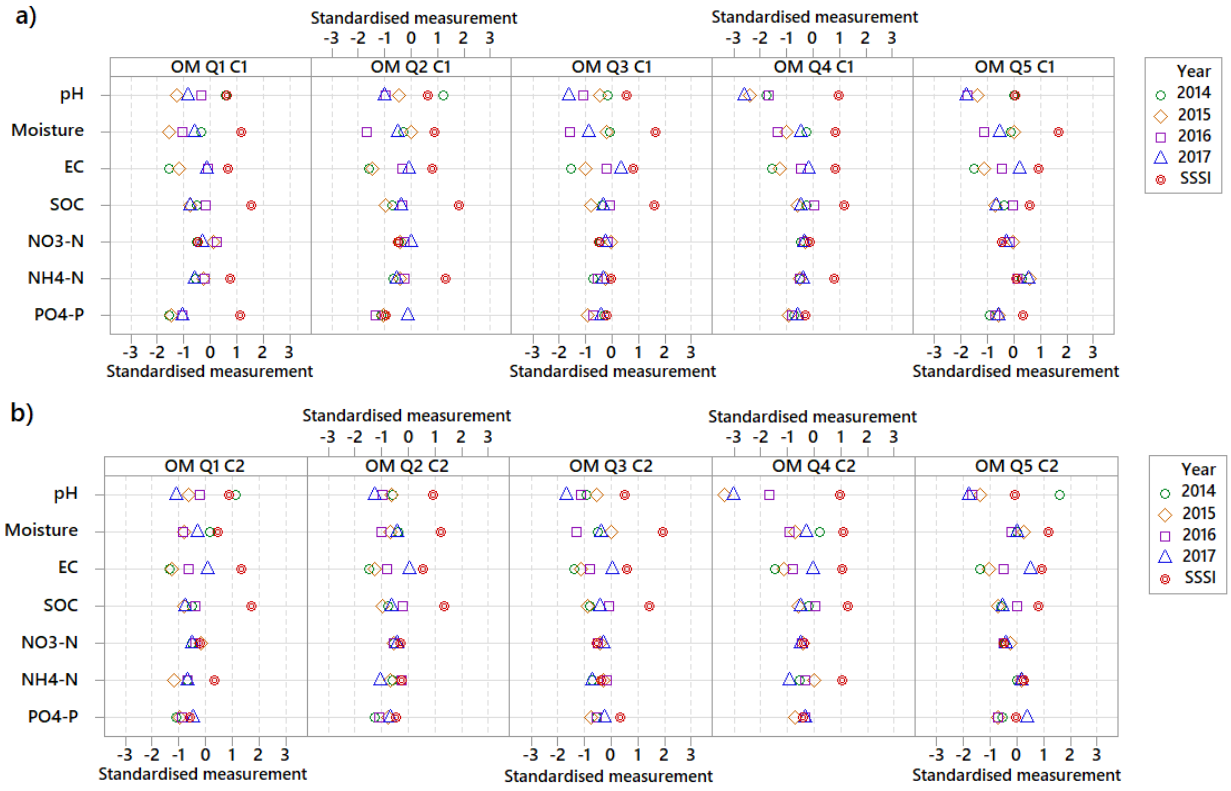


Figure 3.15 Individual value plots of soil variables by year (2014-2017) in a) OM C1 cores, and b) OM C2 cores. Individual values are based on standardised measurements because soil variables had different measurement scales.

There was a significant difference in all soil variables between years, apart from NO_3-N and NH_4-N levels in C2 cores. In C1 cores, pH dropped post-breach and rose to its highest level in 2017. This was a significant change from 2015 and 2016 pH levels, but not from 2014. In C2, pH was lower in 2017 than it was in 2014. Differences in % moisture was significantly different between years in C1 ($p < 0.001$) and C2 ($p = 0.023$) cores. Moisture dropped in 2015 from pre-beach samples, and dropped again in 2016, before rising significantly in 2017. A similar pattern was seen in C2 cores, but it did not reach higher moisture levels than pre-breach samples in 2017. EC changed significantly in C1 ($p < 0.001$) and C2 ($p < 0.001$) cores, with salt levels rising significantly each year in C1. There was also a rise in C2, but this was more gradual, because EC was not significantly different in 2015 from pre-breach data. % SOC changed significantly between years in C1 ($p < 0.001$) and C2 ($p < 0.001$) cores. Carbon levels in C1 did appear to drop after the breach, but this was not a significant change, but they rose to significantly higher levels by 2017. In C2 cores % SOC rose significantly by 2016 but dropped to pre-breach levels in 2017. NO_3-N and NH_4-N levels changed significantly between years in C1 cores (both $p = 0.011$). Levels of the former increased significantly after the breach but dropped substantially in 2017 to levels that were

not a significant change from 2014. NH_4-N was similar in the first two years post breach but rose significantly by 2017. PO_4-P was significantly different between years in C1 ($p = 0.016$) and C2 ($p = 0.042$). Data analysed in both cores had similar patterns, with levels being comparable in 2014, 2015 and 2016, but rising significantly in 2017.

Table 3.7 Differences in soil variables between years (2014-2017) on OM in C1 (top 10cm) and C2 (10-20cm), n = 5 for each variable and each core depth. * = one-way ANOVA using Tukey post hoc-tests if significant differences were detected; ** = Kruskal-Wallis test; ^{ns} = not significant. Average = Mean ± SEM for ANOVA or Median (Mdn) for Kruskal-Wallis test. In post-hoc grouping column, means or medians in each core depth that do not share letters are significantly different (< 0.05), ns = not significant; ◊ = Fisher's pairwise comparisons post-hoc was used because the Tukey method was too conservative to detect differences. Non-standardised data were used for tests because the variables were tested separately by year.

OM - soil variable versus year				Post-hoc analysis				
Soil variable	df	Test statistic	P-value	Year	Average		Grouping	
					C1	C2	C1	C2
Moisture (%)				2014	24.57 ± 0.43	24.60 ± 1.04	b	A
C1 *	3, 16	F = 33.50	< 0.001	2015	21.54 ± 2.87	22.53 ± 1.53	b	a b
C2 *	3, 16	F = 4.21	0.023	2016	14.04 ± 1.11	19.12 ± 1.26	c	B
				2017	38.10 ± 1.56	23.35 ± 0.48	a	a b
pH				2014	7.25 ± 0.10	7.33 ± 0.09	a b	A
C1 *	3, 16	F = 6.28	0.005	2015	7.01 ± 0.07	7.03 ± 0.09	b	B
C2 *	3, 16	F = 5.09	0.012	2016	7.02 ± 0.05	7.06 ± 0.05	b	a b
				2017	7.37 ± 0.03	6.95 ± 0.06	a	B
EC (dS/m)				2014	0.31 ± 0.03	0.47 ± 0.06	d	C
C1 *	3, 16	F = 354.45	< 0.001	2015	1.13 ± 0.18	0.97 ± 0.08	c	C
C2 *	3, 16	F = 122.34	< 0.001	2016	3.36 ± 0.19	1.97 ± 0.12	b	B
				2017	6.10 ± 0.08	3.71 ± 0.21	a	A
SOC (%)				2014	3.64 ± 0.13	3.28 ± 0.17	b c	B
C1 *	3, 16	F = 58.37	< 0.001	2015	3.06 ± 0.10	2.89 ± 0.10	c	B
C2 *	3, 16	F = 14.52	< 0.001	2016	4.24 ± 0.10	4.02 ± 0.11	b	A
				2017	6.92 ± 0.40	3.28 ± 0.09	a	B
NO_3-N (ppm)				2014	0.70 ± 0.44	n/a	b	n/a
C1 *	3, 16	F = 5.14	0.011	2015	16.6 ± 4.70	n/a	a	n/a
C2 **	3	H = 3.76	0.289 ^{ns}	2016	17.00 ± 4.97	n/a	a	n/a
				2017	3.60 ± 3.12	n/a	a b	n/a
NH_4-N (ppm)				2014	57.00 ± 7.76	n/a	b	n/a
C1 *	3, 16	F = 5.13	0.011	2015	68.00 ± 8.34	n/a	a b	n/a
C2 *	3, 16	F = 0.84	0.491 ^{ns}	2016	63.50 ± 5.16	n/a	b	n/a
				2017	99.50 ± 11.0	n/a	a	n/a
PO_4-P (ppm)				2014	14.00 ± 2.00	12.80 ± 1.59	b	b ◊
C1 *	3, 16	F = 4.68	0.016	2015	13.40 ± 1.29	12.60 ± 12.60	b	b ◊
C2 *	3, 16	F = 3.45	0.042	2016	14.00 ± 1.14	13.40 ± 13.40	b	b ◊
				2017	23.00 ± 3.35	17.80 ± 17.80	a	a ◊

3.2.4 Cluster analysis and ordination

3.2.4.1 C1 cores

When all C1 soil variables were analysed together, it was found that SSSI quadrats were grouped together when analysed with August 2014 C1 cores. This was expected, because of the stark differences between the SSSI saltmarsh system and the agricultural

starting state of Steart Marsh and OM. However, in 2014 some of the Site A quadrats were grouped differently to other plots (Figure 3.16 2014).

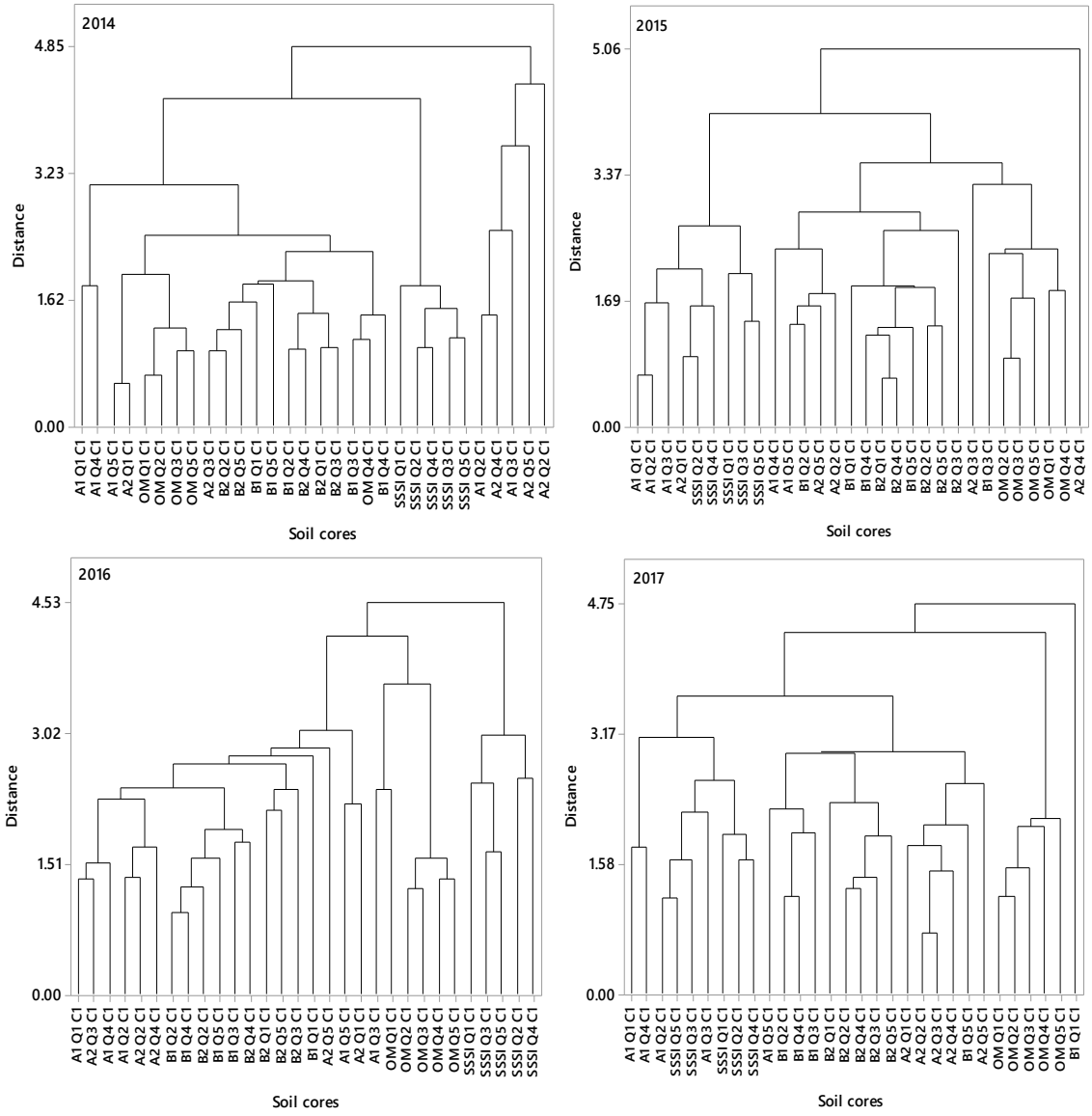


Figure 3.16 Cluster analysis dendrograms of C1 (10cm) soil cores in August 2014-2017. SSSI data are from August 2016. Dendrograms created using the average linkage method and Euclidean distance.

2015 clustering shows that some of the Site A quadrats had moved closer in distance to the SSSI quadrats, and OM were the furthest apart, apart from A2Q4C1 which was dissimilar to all other quadrats (Figure 3.15 2016). The trend that had appeared to develop in 2015 did not continue in August 2016. The SSSI quadrats were on a different cluster than the other plots and were not as similar to the Site A plots as they had been in 2015 (Figure 3.15 2016). However, in 2017 the SSSI quadrats were again more like some of the Site A quadrats. For example, SSSI Q5 C1 was more similar to A1 Q2 C1 than it was to the other SSSI quadrats (Figure 3.16 2017).

Ordination shows the soil variables that were responsible for these groupings. The 2014 ordination (Figure 3.17 2014) shows that the SSSI quadrat grouping was primarily governed by soil moisture, EC and SOC. The Site A quadrats that were grouped had higher levels of NO_3-N and NH_4-N. pH and PO_4-P were the variables responsible for the grouping of the other Steart Marsh and OM quadrats. Groupings in 2015 had changed from initial pre-breach data, and this was primarily governed by the increasing % soil moisture, % SOC and EC in some of the Steart Marsh quadrats, particularly A2 Q1 C1, which was like SSSI Q2 C1. This was a major change in properties in C1 cores. The separation of A2 Q4 C1 from other quadrats was driven by the high pH in this quadrat (Figure 3.17 2015). The pattern did not continue in 2017, but this is mainly because EC was higher in the Steart Marsh plots than it was on the SSSI, which explains why the SSSI was dissimilar to these plots by 2016. OM was very different at this stage than all other plots on the first axis (Figure 3.17 2016). Although EC was still higher on the Steart Marsh plots than the SSSI in August 2017, SOC and NH_4-N was driving similarities between some of the Site A quadrats and the SSSI. OM was still very different at this stage, which can be seen in the dendrogram (Figure 3.16 2017) and ordination (Figure 3.17 2017).

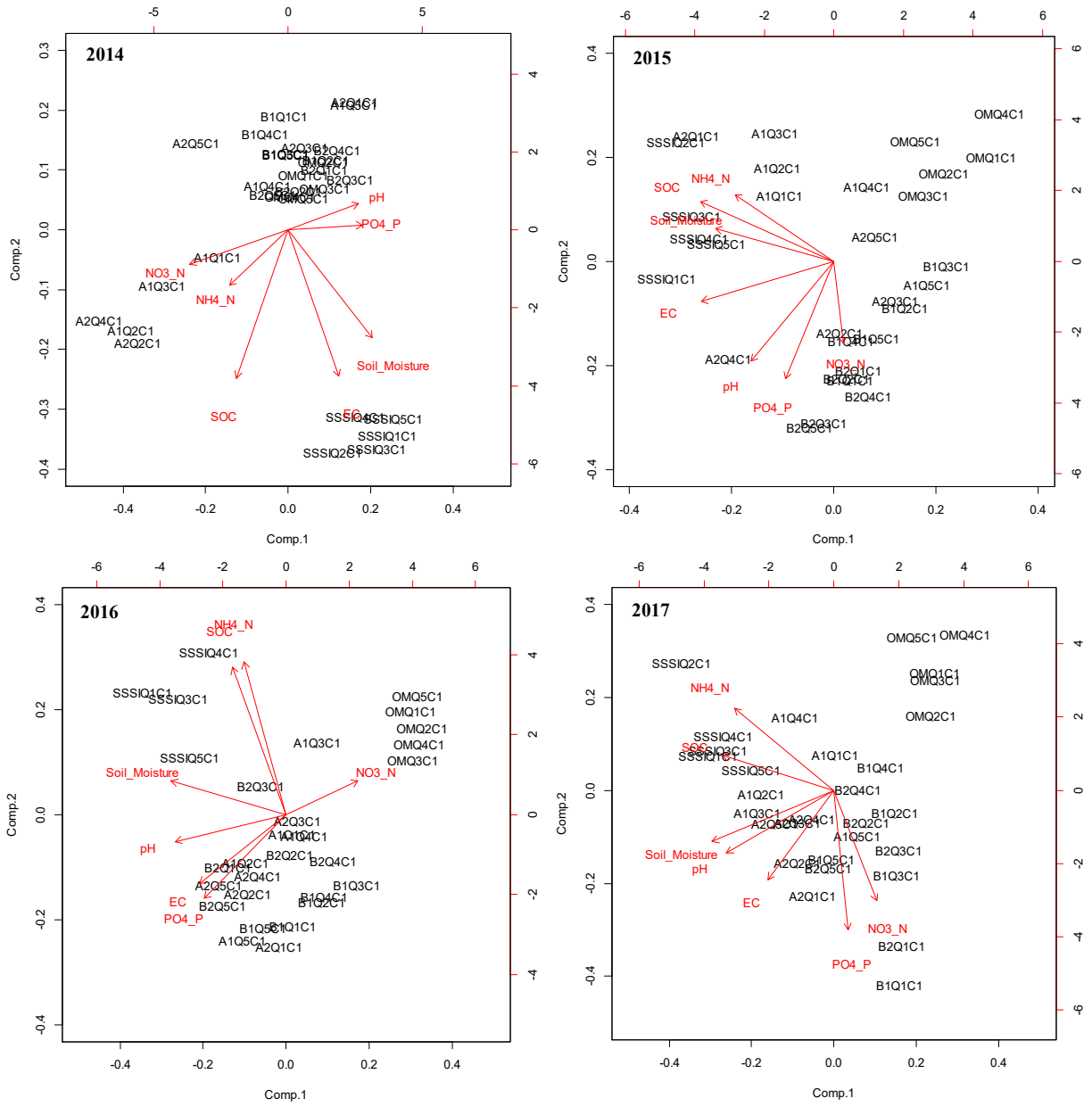


Figure 3.17 Principal Component Analysis biplots of C1 (10cm) soil core groupings in relation to soil variables from August 2014-2017. SSSI data are from August 2016. Both component 1 and component 2 axes are meaningful on all biplots (> 1) and account for 66% of the variation in 2014, 66% in 2015, 65% in 2016 and 69% in 2017.

A Procrustes plot shows how the soil variables have changed in C1 cores in each quadrat from August 2014-2017 (Figure 3.18a and b). In general, Site A quadrats moved toward the SSSI quadrats, and Site B moved in the opposite direction, although this is not true of all quadrats. OM quadrats moved further away from the SSSI quadrats on the first dimension and second dimension.

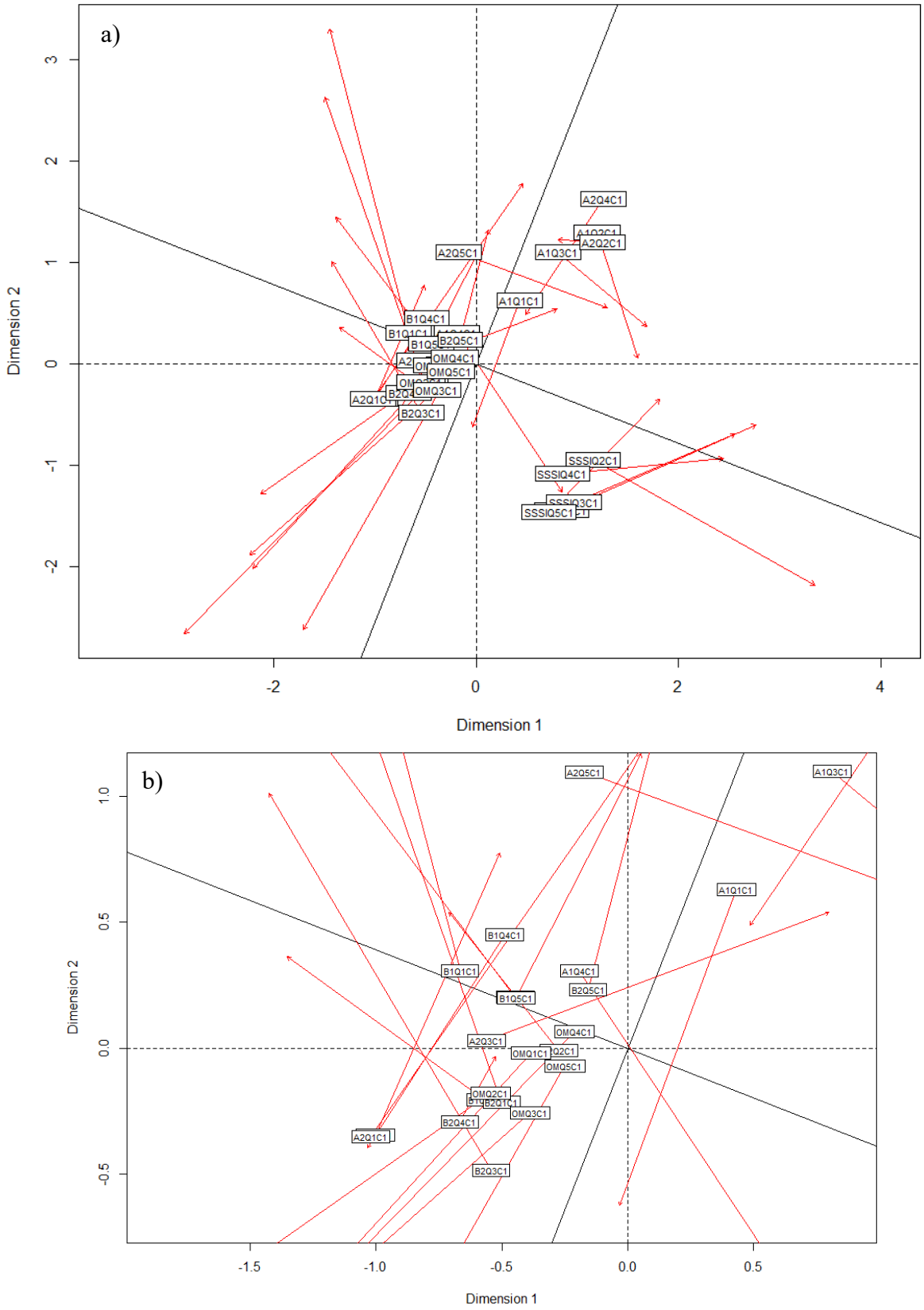


Figure 3.18 A comparison of PCA ordinations of C1 soil cores in August 2014 and 2017; a) Procrustes plot of 2014 and 2017 ordinations; b) magnified image of the plot. Quadrat labels are the position of quadrats in 2014 and arrow heads are the position of quadrats in 2017. The SSSI data was the same in both ordinations (recorded in August 2016).

3.2.4.2 C2 cores

Cluster analysis of C2 core groupings showed a similar pattern to C1 cores in August 2014 (Figure 3.19). The SSSI quadrats were grouped together as expected. The Steart Marsh and OM quadrats were generally more closely grouped, but A2Q2C2 and A2Q4C2 were grouped separately and were dissimilar to other plots. The SSSI quadrats were still grouped separately in 2015. Most of the Site A quadrats were grouped together, which was also the case for Site B quadrats and OM. In 2016 there was a change with the SSSI quadrats in closer distance to the Site A quadrats. At this stage, SSSIQ5C2 was clustered with A2Q4C2. OM quadrats were clustered separately from all other plots in 2016. In 2017, the groupings were similar to 2016, but A1Q2C2 and A2Q4C2 were clustered separately and were distant to the other sites. This is surprising considering that the latter was grouped with a SSSI quadrat the previous year.

Ordination (Figure 3.20) shows that like C1 cores, the SSSI grouping was mainly governed by moisture, EC and SOC. A2Q2C2 and A2Q4C2 were grouped separately because of the relatively high levels of NO_3-N and NH_4-N in the former and NH_4-N in the latter. PO_3-P and pH were governing the groupings of the other Steart Marsh and OM quadrats. There was a similar pattern in 2015, but % SOC levels were drawing some A1 quadrats (particularly A1Q1 and A1Q2) toward the SSSI quadrats. By 2016 OM was dissimilar to all other plots as indicated in the cluster analysis dendrogram (Figure 3.19), but ordination shows that these quadrats were closer to some of the B1 quadrats and the separation from other quadrats was mainly on the second ordination gradient (component 2). Most of the Site B quadrats and some of the Site A quadrats were governed by PO_3-P levels, but other Site A quadrats were grouping in relation to NO_3-N and NH_4-N and % SOC. These quadrats were closer on the gradient to the SSSI than in the previous year post-breach. By 2017, most of the Site A quadrats were closer to the SSSI, although levels of some soil variables had higher measurements than found on the target state. OM was grouped separately, and most of the Site B quadrats were grouping on the second gradient due to high levels of PO_3-P in 2017.

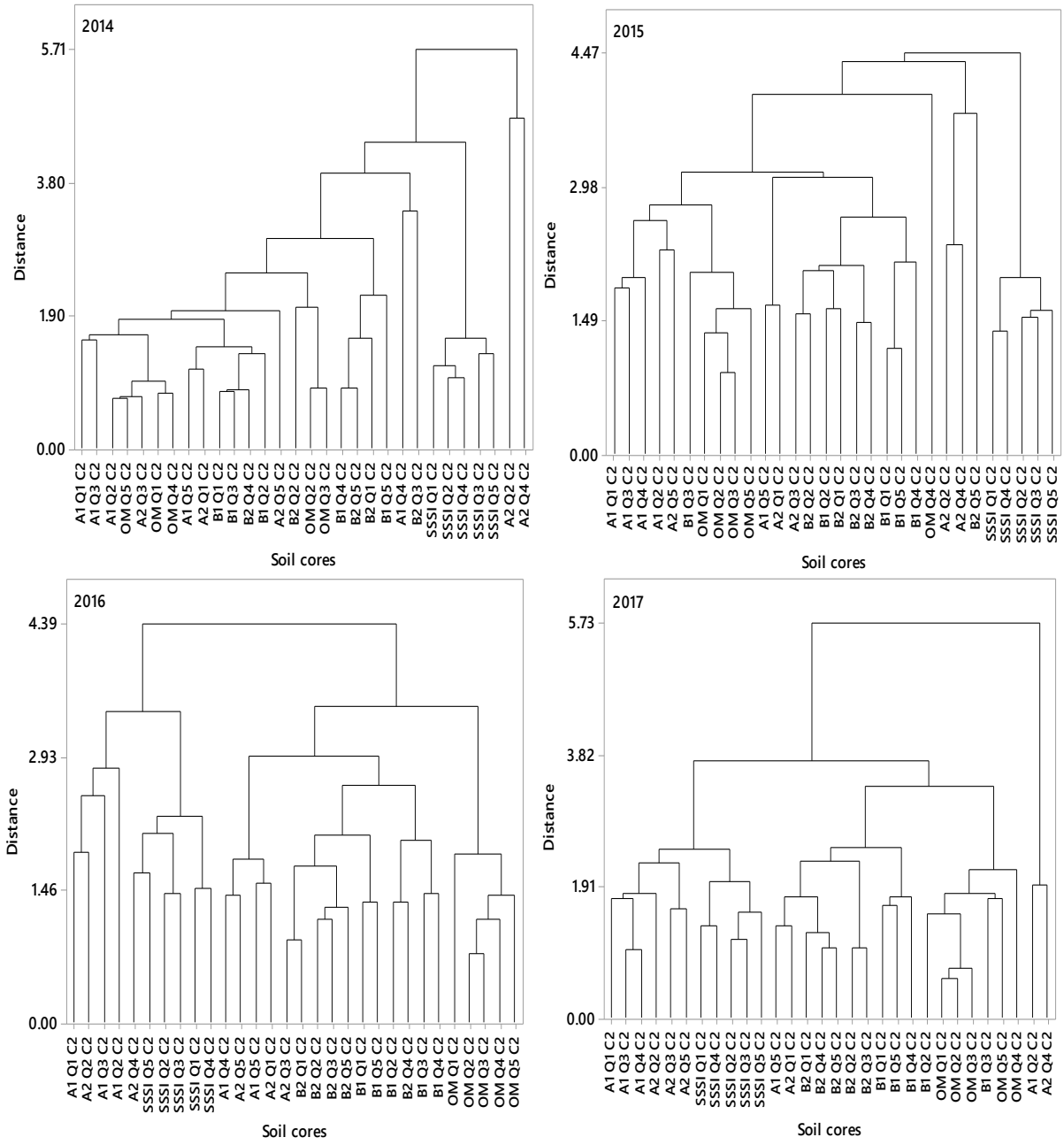


Figure 3.19 Cluster analysis dendrograms of C2 (10-20cm) soil cores from August 2014-2017. SSSI data are from August 2016. Dendrogram created using the average linkage method and Euclidean distance.

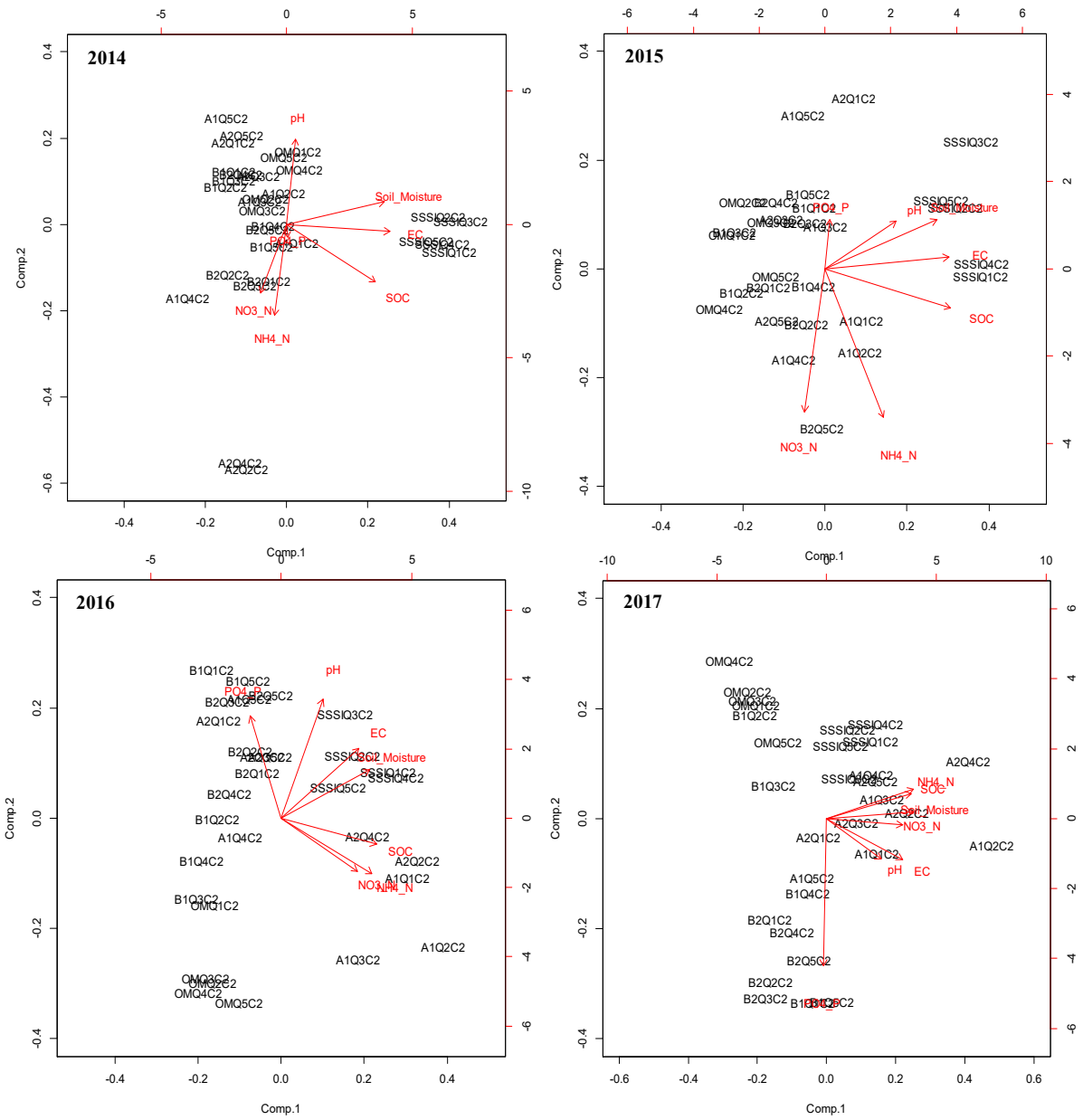


Figure 3.20 Principal Component Analysis biplots of C2 (10-20 cm) soil core groupings in relation to soil variables from August 2014-2017. SSSI data are from August 2016. Both component 1 and component 2 axes are meaningful on all biplots (> 1) and account for 59% of the variation in 2014, 58% in 2015, 68% in 2016 and 73% in 2017.

A Procrustes plot shows how the soil variables changed in C2 cores in each quadrat from August 2014-2017 (Figure 3.21a and b), and the result is similar to the C1 cores. Most Site A quadrats moved toward the SSSI quadrats, and Site B moved in the opposite direction, but, like C1 cores, this is not true of all quadrats. Similarly to C1 cores, OM quadrats moved further away from the SSSI quadrats on the first dimension.

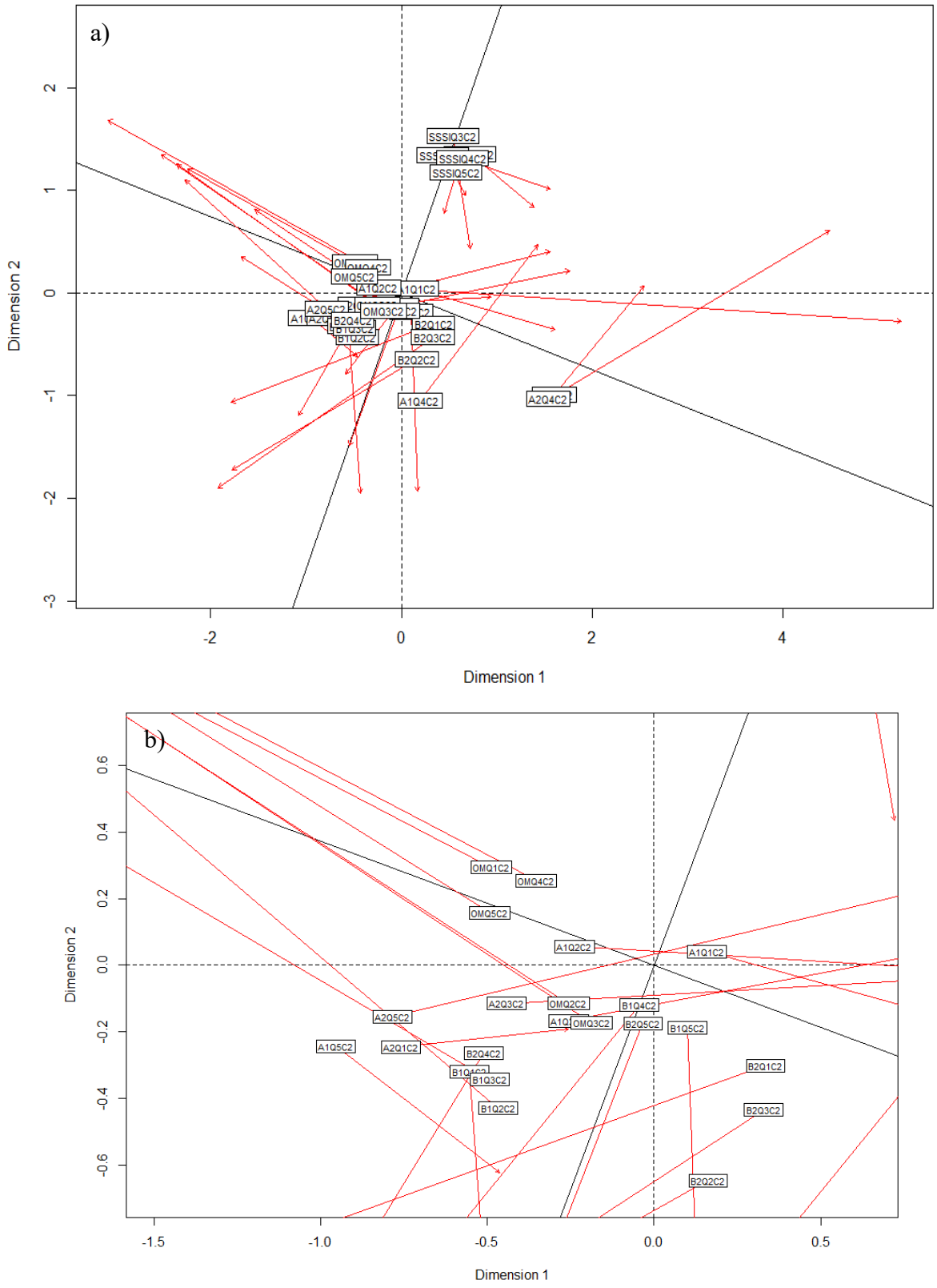


Figure 3.21 A comparison of PCA ordinations of C2 soil cores in August 2014 and 2017; a) Procrustes plot of 2014 and 2017 ordinations; b) magnified image of the plot. Quadrat labels are the position of quadrats in 2014 and arrow heads are the position of quadrats in 2017. The SSSI data was the same in both ordinations (recorded in August 2016).

3.3 Soil Organic Carbon – tons of carbon per hectare (tC/ha)

SOC tC/ha is included as an estimate of the storage of SOC in the soil each year in comparison to the SSSI target state. This was based on estimated bulk density (Table 3.8).

Table 3.8 Estimation of the bulk density of soil cores (C1 and C2) each year (2014-2017).

Year	Core	Plot	N	Mean	SE Mean (±)	Core	Plot	N	Mean	SE Mean (±)
2014	C1	A1	5	0.50	0.05	C2	A1	5	0.50	0.03
		A2	5	0.52	0.06		A2	5	0.54	0.02
		B1	5	0.54	0.02		B1	5	0.57	0.01
		B2	5	0.56	0.02		B2	5	0.62	0.02
		OM	5	0.52	0.01		OM	5	0.63	0.01
		SSSI	5	0.46	0.05		SSSI	5	0.58	0.01
2015	C1	A1	5	0.55	0.07	C2	A1	5	0.66	0.03
		A2	5	0.55	0.05		A2	5	0.73	0.04
		B1	5	0.50	0.03		B1	5	0.55	0.04
		B2	5	0.47	0.02		B2	5	0.62	0.04
		OM	5	0.57	0.04		OM	5	0.65	0.03
		SSSI	5	0.46	0.05		SSSI	5	0.58	0.01
2016	C1	A1	5	0.41	0.03	C2	A1	5	0.51	0.03
		A2	5	0.48	0.04		A2	5	0.56	0.05
		B1	5	0.47	0.04		B1	5	0.57	0.04
		B2	5	0.47	0.01		B2	5	0.55	0.04
		OM	5	0.53	0.01		OM	5	0.57	0.03
		SSSI	5	0.46	0.05		SSSI	5	0.58	0.01
2017	C1	A1	5	0.52	0.03	C2	A1	5	0.55	0.03
		A2	5	0.52	0.02		A2	5	0.61	0.06
		B1	5	0.51	0.03		B1	5	0.69	0.05
		B2	5	0.52	0.01		B2	5	0.62	0.02
		OM	5	0.55	0.05		OM	5	0.52	0.02
		SSSI	5	0.46	0.05		SSSI	5	0.58	0.01

3.3.1 C1 cores

In 2014 a difference in SOC tC/ha was not detected between plots in C1 cores (one-way ANOVA, $F(5, 24) = 2.07$, $p = 0.105$). There was a difference between plots in 2015 (one-way ANOVA, $F(5, 24) = 7.50$, $p < 0.001$) with Tukey post-hoc tests showing that the SSSI had significantly higher SOC tC/ha than plots B1, B2 and OM, but not A1 and A2. However, A1 was not different than the Site B plots and OM, and A2 was not different than the Site B plots. A Welch's Test was used to test differences in SOC tC/ha between plots in 2016 because equal variances could not be assumed (Levene's Test, $p = 0.010$). A difference

was detected (F (5) = 13.98, p <0.001), and Games-Howell pairwise comparisons showed that A1 had significantly lower SOC tC/ha than plots B2, OM and the SSSI. Although plots A2 and B2 were not significantly lower than B2, OM and the SSSI, a difference could not be detected between these plots and A1. There was a significant difference between plots in 2017 (F (5, 24) = 3.96, p = 0.009). The SSSI did not have significantly higher SOC tC/ha than plots A1, A2 and B1, but a difference was detected between the SSSI and B2 and OM. However, plots A1, A2 and B1 were not significantly higher in SOC tC/ha than B2 and OM (Figure 3.22a).

3.3.2 C2 cores

In 2014 a difference in SOC tC/ha was detected between plots in C1 cores (one-way ANOVA, F (5, 24) = 9.97, p <0.001). The SSSI had SOC tC/ha significantly higher than all other plots. There was a difference between plots in 2015 (one-way ANOVA, F (5, 24) = 7.28, p <0.001) with Tukey post-hoc tests showing that the SSSI had significantly higher SOC tC/ha than plots B1, B2 and OM, but not A1 and A2, which is similar to C1 cores. Like C1 cores, A1 was not different to the Site B plots and OM, and A2 was not different to the Site B plots, but it was not different to OM in C2 cores. A difference was detected in 2016 (F (5, 24) = 7.05, p <0.001), with the SSSI having significantly higher SOC tC/ha than all other plots. There was also a significant difference between plots in 2017 (F (5, 24) = 15.03, p < 0.001). Like C1 cores, the SSSI did not have significantly higher SOC tC/ha than plots A1, A2 and B1 in 2017, but a difference was detected between the SSSI and B2 and OM. Unlike C1 cores, plots A1, A2 and B1 were significantly higher in SOC tC/ha than B2 and OM (Figure 3.22b).

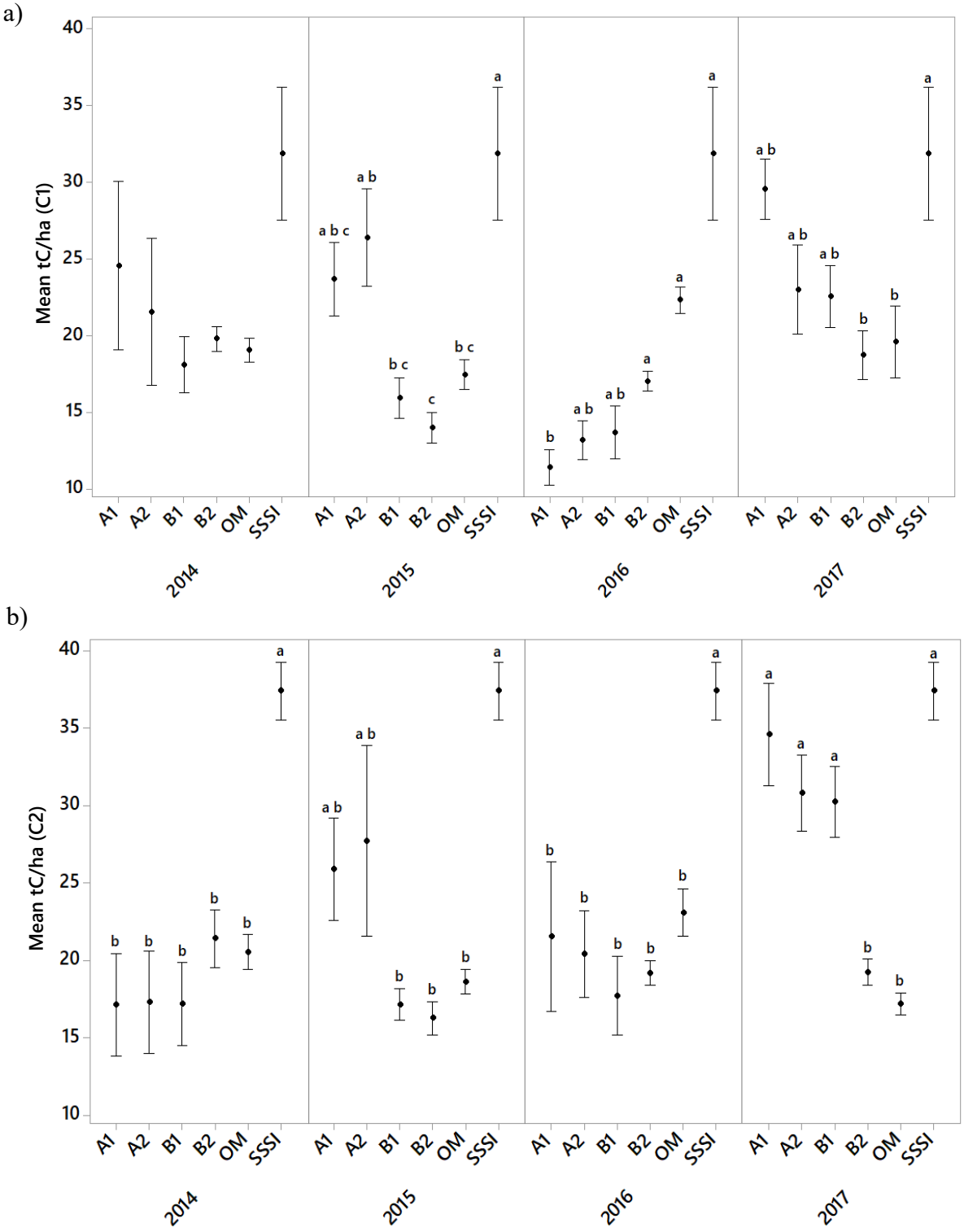


Figure 3.22 Estimate of average SOC tC/ha in each plot each year in a) C1 cores (n = 5 in each plot each year) and b) C2 cores (n = 5). Each year in each core has been analysed separately. In each year means that do not share letters are significantly different (< 0.05). If there are no letters, there are no differences between plots that year.

3.4 Accretion and compaction

Prior to the breach in September 2014, levels of each quadrat were taken to ensure comparability of plot height. It was originally planned that posts would be used to measure

accretion, but due to concerns with the posts on the sites being damaged by grazing animals, the automatic level was used again in 2016 and 2018 to gain accurate elevation data of each quadrat. This provides a detailed comparison with the 2014 levels (Table 3.9).

Table 3.9 Mean AOD of each quadrat on Steart Marsh and Otterhampton Marsh in 2014 (pre-breach starting state), 2016 (2-years post-breach) and 2018 (4-years post-breach). Data for each quadrat are averaged from five sample points. SEM = Standard Error of the Mean.

Quadrat	Elevation 2014 (metres) ± SEM	Elevation 2016 (metres) ± SEM	Elevation 2018 (metres) ± SEM
A1 Q1	5.738 ± 0.005	5.827 ± 0.005	5.895 ± 0.007
A1 Q2	5.556 ± 0.031	5.635 ± 0.039	5.736 ± 0.027
A1 Q3	5.879 ± 0.011	5.960 ± 0.009	6.010 ± 0.010
A1 Q4	5.784 ± 0.018	5.871 ± 0.007	5.959 ± 0.007
A1 Q5	5.925 ± 0.007	5.952 ± 0.008	5.990 ± 0.009
A2 Q1	5.681 ± 0.008	5.827 ± 0.004	5.905 ± 0.009
A2 Q2	5.691 ± 0.021	5.833 ± 0.018	5.915 ± 0.020
A2 Q3	5.819 ± 0.020	5.945 ± 0.016	6.012 ± 0.016
A2 Q4	5.735 ± 0.015	5.892 ± 0.015	5.963 ± 0.023
A2 Q5	5.576 ± 0.013	5.736 ± 0.005	5.786 ± 0.005
B1 Q1	5.834 ± 0.013	5.884 ± 0.008	5.950 ± 0.011
B1 Q2	5.945 ± 0.015	5.971 ± 0.013	6.021 ± 0.010
B1 Q3	5.976 ± 0.008	6.002 ± 0.005	6.047 ± 0.009
B1 Q4	5.821 ± 0.005	5.875 ± 0.004	5.954 ± 0.003
B1 Q5	5.777 ± 0.005	5.857 ± 0.006	5.927 ± 0.006
B2 Q1	5.883 ± 0.012	5.915 ± 0.008	5.981 ± 0.009
B2 Q2	5.891 ± 0.013	5.922 ± 0.014	5.984 ± 0.005
B2 Q3	5.807 ± 0.005	5.872 ± 0.004	5.959 ± 0.004
B2 Q4	5.914 ± 0.007	5.937 ± 0.002	6.015 ± 0.007
B2 Q5	5.816 ± 0.004	5.875 ± 0.006	5.952 ± 0.007
OM Q1	5.532 ± 0.003	5.517 ± 0.006	5.504 ± 0.001
OM Q2	5.539 ± 0.002	5.537 ± 0.004	5.519 ± 0.002
OM Q3	5.527 ± 0.004	5.522 ± 0.005	5.513 ± 0.006
OM Q4	5.546 ± 0.004	5.538 ± 0.005	5.502 ± 0.002
OM Q5	5.489 ± 0.007	5.482 ± 0.005	5.478 ± 0.004

In general, data from the posts and the automatic level showed that sites on the scheme were accreting, although there was erosion on OM. This is highlighted in the mean accretion/erosion in each quadrat from 2014-2018, which were significantly different to one another (one-way ANOVA, $F(24, 100) = 92.15$, $p<0.001$). Tukey pairwise comparisons showed that the differences were between the A2 quadrats and the B1, B2, and OM quadrats. A1 quadrats had similarities with A2 plots and B1 and B2 plots, with quadrats A1Q3 and A1Q5 having similarities with the Site B plots rather than A2 plots. OM quadrats were different than all other plots because these had eroded (Figure 3.23). These data indicate that A2 plots accreted the most over the study period, but Site B plots were higher in the tidal frame throughout the study period.

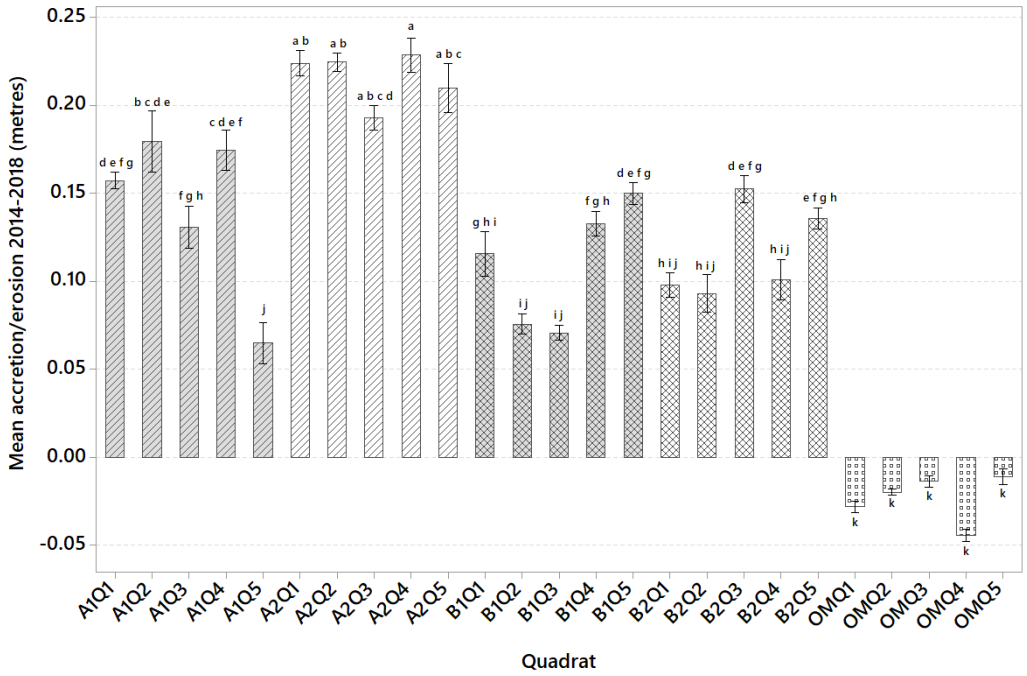


Figure 3.23 Mean accretion/erosion in each quadrat from 2014-2018 (n = 5 in each quadrat). Error Bars represent Standard Error of the Mean. Means with different letters are significantly different from one another.

Despite the rapid accretion on A2, both Site A plots were at similar heights (c.5.9 metres AOD on average) by the end of the study. A similar pattern occurred with Site B plots which both averaged at just below 6 metres AOD by 2018 (Figure 3.24).

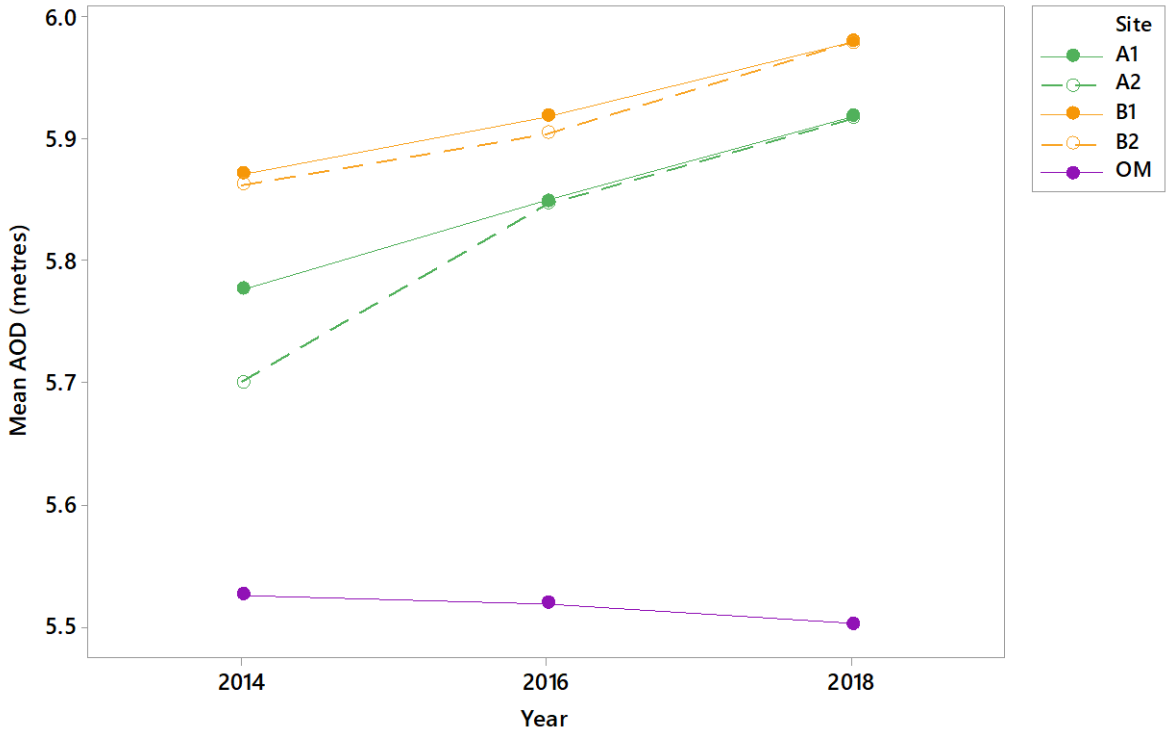


Figure 3.24 Mean AOD (metres) in each quadrat from 2014-2018.

Compaction data were tested with a one-way ANOVA using Welch's Test because equal variances could not be assumed. A significant difference between sites was detected at 200psi ($F(4) = 91.69$, $p < 0.001$; equal variances not assumed) and at > 300psi ($F(4) = 63.80$, $p < 0.001$; equal variances not assumed). As equal variances could not be assumed, a Games-Howell post-hoc test was used on both datasets. It was discovered that at 200psi, A1 (M = 0.289 m below surface ± 0.015 SEM), A2 (M = 0.344 m ± 0.012) and OM (M = 0.127m ± 0.005) were significantly different from one another and the Site B plots. No difference was detected between B1 (M = 0.201 m ± 0.010) and B2 (M = 0.210 m ± 0.008). At > 300psi A1 (M = 0.448m ± 0.022) and A2 (M = 0.501m ± 0.014) were not different from one another, but they were different to all other plots. B1 (M = 0.262m ± 0.011) was different to all other plots, and B2 (M = 0.322m, ± 0.016) and OM (M = 0.255m ± 0.010) were not significantly different from one another (Figure 3.25).

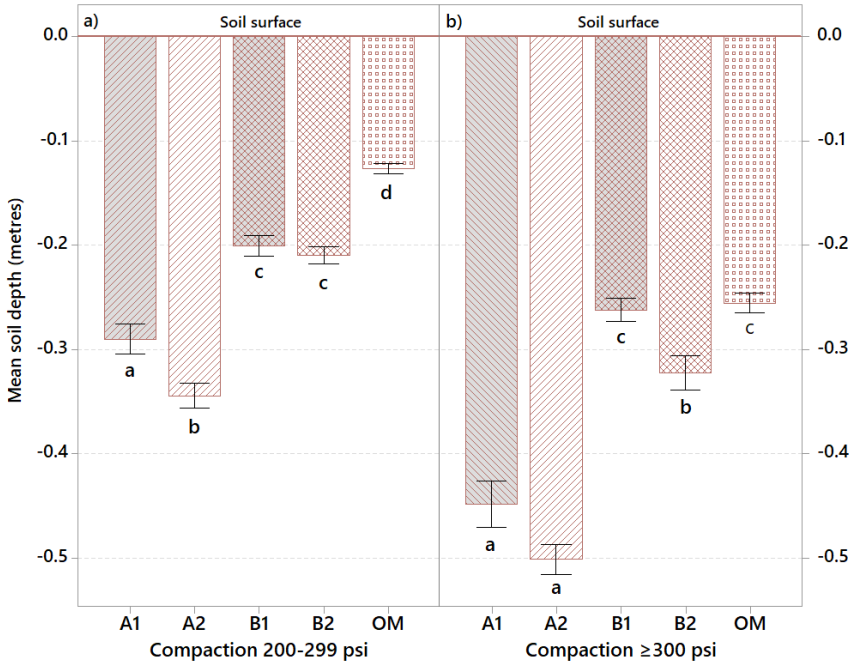


Figure 3.25 Soil compaction depths on study plots in August 2018 at a) 200-299 psi (n = 25 for each plot) and b) > 300 psi (n = 25), measured using a Dickey-John soil compaction tester with a ½ tip. Minus values on y-axis are used to show mean depth below soil surface when tester hit level of compaction. Error bars represent Standard Error of the Mean. In each psi category, means that do not share a letter are significantly different. SSSI is not included because compaction was not detected.

To estimate the levels of compaction of the soil before the breach in August 2014, accretion/erosion data from 2014-2018 were subtracted from 2018 compaction data. This acts as an estimate of initial compaction because it does not consider any subsequent compaction that could have happened between these times or any erosion that was not

detected between elevation sampling. However, it does provide a good approximation of how the sites compared in terms of compaction pre-breach.

These data were tested with a one-way ANOVA using Welsh's Test. A significant difference between sites was detected at 200psi ($F(4) = 10.99$, $p < 0.001$; equal variances not assumed) and at > 300psi ($F(4) = 31.03$, $p < 0.001$; equal variances not assumed). A Games-Howell post-hoc test was used on both datasets. It was discovered that at 200-299 psi, A1 (M = 0.149m below surface ± 0.016 SEM), A2 (M = 0.128m ± 0.012), and OM (M = 0.150m ± 0.010) were not significantly different to one another, but A2 was not significantly different to B1 (M = 0.092 ± 0.008) and B2 (M = 0.094 ± 0.010). However, A1 and OM were significantly different to both Site B plots. At > 300psi A1 (M = 0.307m ± 0.026) and A2 (M = 0.285m ± 0.014) were not different from one another or OM (M = 0.279m ± 0.008), but they were different to B1 (M = 0.153m ± 0.010) and B2 (M = 0.206m, ± 0.017), which were not different from one another (Figure 3.26).

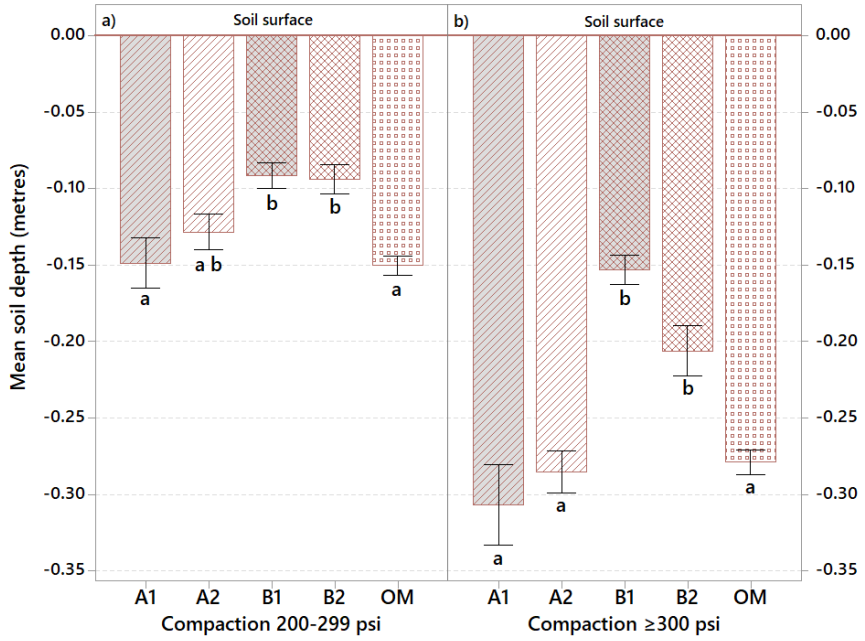


Figure 3.26 Estimated soil compaction depths on study plots in August 2014 at a) 200-299 psi (n = 25 for each plot) and b) > 300 psi (n = 25). Data are derived from subtracting accretion/erosion levels (between 2014-2018) from August 2018 compaction data. Minus values on y-axis are used to show mean depth below soil surface when tester hit level of compaction. Error bars represent Standard Error of the Mean. In each psi category, means that do not share a letter are significantly different. SSSI is not included because compaction was not detected.

3.5 Discussion

The aim of this chapter was to assess soil characteristics within permanent quadrats on Steart Marsh and compare these to the managed OM site and the SSSI target state in space (between plots) and time (from 2014-2017). It was important to monitor key soil variables to assess how these were changing over time after the initial breach, because the soil supports halophytic flora and invertebrate fauna succession. Discussion of how soil characteristics relate to vegetation succession and invertebrate colonisation is in Chapter 6.

The key findings in this chapter are that moisture and %SOC measurements in soil on Site A plots had the most comparable measurements to the target state by August 2017 (in comparison to all other plots). pH and salinity were similar across the study plots, but both of these soil characteristics had lower measurements on OM. Nitrate-Nitrogen and Ammonium-Nitrogen had a high level of variation pre-breach, but this variation reduced post-breach. Phosphate-phosphorus measurements were higher on the Site B plots by the end of the study in comparison to the other plots, including the SSSI. On Plots A1 and A2 moisture increased post-breach, and there was an increase in salinity. pH did not change and there was no detectable change in nitrate-nitrogen and ammonium-nitrogen. There was an increase in phosphate-phosphorus, with the highest levels measured in 2017. On plots B1 and B2 moisture increased post-breach (but soil moisture was lower than on Site A plots). Salinity rose to higher levels after tidal inundation, but pH remained similar in these plots, although it was marginally lower post-breach in C2 cores. A pattern did not emerge with nitrate-nitrogen and ammonium nitrogen over the course of the study on Site B plots, but phosphate-phosphorus did increase and was substantially higher in C1 and C2 cores by August 2017, and this was higher than measurements on the target state. Measurements of all soil variables were lower on OM than on all other plots from August 2015, but soil moisture did increase in 2017 after dropping to its lowest levels in 2016. pH was more alkaline in August 2017 C1 cores in comparison to 2014 levels, but it was more acidic in C2 cores. SOC did increase in C1 cores on OM, but it was similar in C2 cores. Ammonium-nitrogen and phosphate-phosphorus were at their highest levels in 2017 on OM, but there was no pattern with nitrate-nitrogen measurements. In terms of soil characteristics Site A plots were the most similar to the SSSI target state by the end of the study and these changes to comparable levels were more rapid on this site. All plots apart from OM accreted over the course of the study but plot A1 accreted the most. It is likely that compaction on Site B and OM plots may have had an influence on differences in soil characteristics in comparison to Site A plots and the SSSI.

The starting states in this study had not been under a tidal regime so it was expected that soil moisture would be lower than on the SSSI. This was the case, with soil moisture in the top 10cm of soil on the SSSI averaging > 38%. OM had an average of almost 25%, which was the highest out of the studied plots. This may have been because water had been let in on this site before the breach on Steart Marsh. B2 had the highest soil moisture content on the Steart Marsh plots in August 2014. In soil 10-20cm deep, average moisture % was higher in most of the plots, but it was lower in B2 and the SSSI. By the end of the study, soil moisture in the top 10cm had increased to levels that were almost the same as the SSSI on the Site A plots (36.3% A1; 35.7% A2). Counterintuitively, Soil moisture dropped on both Site B plots after the breach and did not reach higher levels until August 2017. This was not expected, because the tide would have consistently submerged the site throughout each year. However, standing water was on the site for large periods, which suggested that the soil was either saturated or compacted, which would not allow water to filter through it (Nawaz, Bourri and Troland, 2013). Data from this study indicate that it was likely to be the latter.

Due to these observations it was questioned whether compaction was playing a role in succession. In August 2018, Site A plots were the least compacted, which would provide a deeper substrate for plants to root. Although only an approximation could be made regarding compaction pre-breach, it is likely that the Site B plots were the most compacted at the beginning of the study. Soil compaction is a major concern in agriculture and has been shown to reduce crop yield (for review see Nawaz, Bourri and Troland, 2013), and it can also lead to waterlogging on saltmarshes (Kelleway, 2002). The permanent plots were placed in an area of the arable field that was close in AOD to the Site A plots and the SSSI, but land surrounding the plots on Site B were all above 6 metres AOD at the start of the study. Subsequently, water gathered on the Site B plots, and it could not drain due to its lower elevation to the surrounding land on the site. It is also likely that the compacted soil on these plots formed an aquiclude, which meant that water could not percolate through the substrate. It is likely that this lack of permeability had impact on plant growth (see Chapter 4), and it has been evidenced that halophyte survival and colonisation is poorer in plots that are lower in the tidal frame when compared to raised areas (Mossman, Grant and Davy, 2020).

The pH of the site is a key parameter because it influences nutrient uptake in plants, and it was a major factor to assess in relation to plant community development. Data from the UK Soil Observatory map viewer shows that the pH of topsoil on Steart peninsula in 2007 was c.7.05, which had changed from acidic conditions that were recorded in the late 70s and 90s. It was discovered during sampling in August 2014 that some of the sites were

more alkaline than expected, but at the time of the breach eight years had passed since 2007 data were recorded and there was an apparent trend toward alkaline soils. In B1 C2 cores and OM, pH did appear to alter post-breach, but the difference was negligible. There was no detectable difference between years on the other plots. Kadiri *et al.* (2011) found that pH was higher on an MR than a natural saltmarsh, but this does not appear to be the case on Steart as pH was either lower or like the SSSI post-breach. However, soil pH was only tested in August, whereas Kadiri *et al.* (2011) sampled seasonally. Nevertheless, pH readings across sites indicate that the soil was suitable for plants that were adapted to alkaline conditions. Specific plant tolerances will be explored further in Chapter 6.

Plots on Steart Marsh had accreted rapidly, with > 20cm sediment depth on top of underlying land on A2 sites. At the Tollesbury MR scheme, sediment increased by c.15-21cm in a similar marsh zone in approximately 8 years (Reading *et al.*, 2008), although original AOD on Tollesbury was lower than at Steart. At the Paull Holme Strays in the Humber estuary, accretion was rapid on the MR site (Mazik et al., 2010), so it was reasonable to suggest that this could occur at Steart because of the available sediment and tidal range in the Severn Estuary. The rapid accretion meant that it was unlikely that C1 cores on plot A2 contained much or any of the original agricultural soil, so comparisons made in the lower 10-20cm of soil may have been between accreted sediment in some cores and original soil in others. This may make it difficult to determine the effect of agricultural soils as a starting state. Deeper cores could be taken to assess this in future studies.

Salt levels within the soil (measured by electrical conductivity) were very low on Steart Marsh and OM in August 2014, especially in comparison to the SSSI, although significant differences could not be detected between the SSSI and all other plots. It was expected that EC would be lower on Steart Marsh and OM because these sites had not been under a tidal regime. The lack of detectable differences may have been due to limitations of adjusting p-values in post-hoc analysis. After the breach, salt levels inevitably increased in the substrate on Steart Marsh and OM, with the highest levels of salt being in the top 10cm of soil (C1) in all plots. It is important to note that some of the C1 cores would have been sediment that had accreted on the site. Salt levels in the top 10cm of soil rose in plots A2, B1 and B2 to higher levels than the SSSI by August 2016. Kadiri *et al.* (2011) found that salinity was higher on an MR site than a natural saltmarsh, which was attributed to lack of vegetation coverage on the MR and the effect of drying out wet sediment in summer months that leaves accumulated salt in the surface of substrate. This effect helps to explain why salt levels were higher in the topsoil than in soil cores 10-20cm deep. By 2016, the sites (apart

from OM) had continued to accrete and most of the top layer of soil would have been sediment from the estuary, which may also help to explain why this was high in salt, but levels were lower further down the soil profile. The increase in salt may have partly been due to compaction and the lack of vegetation on these sites in comparison to the SSSI, because evidence suggests that these two factors can substantially increase soil salinity (Van Klink *et al.*, 2015). This is because compaction reduces plant growth and vegetation reduces water evaporation, which subsequently slows the passage of salt through soil (Srivastava and Jefferies, 1996). OM did not reach comparable levels to the SSSI throughout the study. This was not unexpected, because it was under brackish, rather than saline conditions.

Average soil organic carbon (SOC) levels in August 2014 were not significantly greater on the Site A plots than on the Site B plots and OM. However, there was considerable variation in the pasture plots compared to the other plots. The SSSI soil (collected in 2016) had a higher average % SOC than all other sites, and there was less variation on this site. The pasture site may have had a lot of variation in % SOC because of the way it had been managed previously. Grazing can alter soil properties due to trampling and excretions by livestock (Wei *et al.*, 2011), and it is possible that excretions made by animals in the past were present in the core samples. This differed from the Site B plots and OM that were in arable crop rotations pre-breach. In comparison, Site B was compacted, so there may have been less carbon possibly because soil compaction decreases the production of vegetation and storage of SOC (Piñeiro *et al.*, 2010).

In August 2015, the top 10cm of soil had decreased in average % SOC on all sites apart from A2, although there was no significant difference when comparing medians. However, there was still more variation on the pasture sites than the other study sites. The Site B plots were in standing water during this period, which meant that there would have been anaerobic conditions, which is not beneficial for plant growth (Boorman, 2003). This, coupled with the compaction on the site and the fact that there was less coverage of vegetation as a starting state (see Chapter 4), may be why there was less % SOC on these sites. The effect of anaerobic conditions may also reduce microbial activity, which will limit the breakdown of organic matter in the soil over the short term (Kadiri et al., 2011). This effect will be apparent on a MR scheme in the initial stages post-breach. However, it is important to consider the accretion on the site, which changes the soil profile. In all circumstances on the pasture site, the top level of the soil in 2016 consisted of sediment that has been brought in by the tidal regime. The sediment would not have contained as much SOC as there was pre-breach in this layer because of the lack of plant matter, and the original

10cm of soil taken in 2014 contained roots from the diverse grasses and ruderals that were on the site. The second layer of the soil (C2 10-20cm) had a higher average % SOC than there was in the second layer in 2014, and this was true of all sites. However, if these data are compared with the original top 10cm of soil in 2014, there was still less % SOC on the pasture sites.

The average % SOC levels in the top 10cm of soil on B2 and OM dropped in 2015 but was higher in 2016 and 2017. The drop in % SOC in 2015 may have been due to the anaerobic conditions that can significantly alter the microbial communities in the soil (Inglett, Reddy and Corstanje, 2005). The vegetation levels were low in 2016 on both sites, so it is perhaps surprising that % SOC increased. However, after post-hoc tests, it was discovered that there was no significant difference between years detected in the top 10cm of soil in B2. Nevertheless, % SOC in OM C1 cores in 2017 was significantly higher than in 2015 and 2016, which could be due to livestock being present on the site since 2015. Livestock excretions can increase organic matter in soil (Wei *et al.*, 2011), and both sheep and cattle had access to the plots each year.

By 2017, there was no detectable difference in % SOC in the top 10cm of soil between the SSSI and A1, or in C2 cores between the SSSI, Site A plots and B1. This suggests that these Steart Marsh plots had similar levels of % SOC as the target state. It is likely that this was influenced by the colonisation of plant species, which is discussed further in Chapter 4 and Chapter 6.

SOC tC/ha was approximated in this study, and SSSI estimates were similar to those predicted by Ford *et al.* (2019) on Welsh saltmarshes. This measure of SOC did show that the effect of tidal inundation reduced estimated SOC levels, particularly by 2016, but rose again in 2017, with plots A1, A2 and B1 having similar levels to the SSSI. The vegetation on Site A plots would have been decomposing after the breach, and it is likely that some of this organic matter would have been in cores in 2015, but C1 levels were much lower in 2016 because it mainly consisted of sediment. Deeper cores could be taken in future sampling to assess how much SOC is buried, although Ford *et al.* (2019) argues that SOC in the surface of mineral sediments is a reliable indicator of SOC stock in British saltmarshes. The estimated SOC stock in some of the study plots by 2017 is encouraging, especially because data are similar to the SSSI.

NO_3-N was very variable on the Site A plots in August 2014. These plots had been a pasture, which would have been grazed by livestock. There were very high spikes of nitrate,

which could have potentially been from animal faeces within the soil. In comparison, the SSSI had very low levels of NO_3-N in the soil, but nitrate moves easily with water (Lamb, Fernandez and Kaiser, 2014), which may be a reason why levels were lower, and the dense vegetation may have been using this nutrient. Nitrate levels were a lot lower on the Site A plots post-breach, but there was not a clear pattern by year, which compares to research by Negrin *et al.* (2011) who did not find a trend in nitrate levels on saltmarshes. Although descriptive data show that median levels of ammonium were low on the Site B plots in C1 cores, concentrations were not significantly lower than in the Site A plots, but they were lower than in SSSI C1 cores. Conversely, in 2017 Site A plots had significantly higher NH_4-N concentrations than the Site B plots and OM in C2 cores. According to Hazelden and Boorman (1999) NH_4-N is likely to be less concentrated in coarse, aerobic soils, which may be why concentrations were higher in the less compacted Site A plots that had not been under standing water. However, a difference in NH_4-N could not be detected between the Site B plots, OM and the SSSI in C2 cores by the end of the study. Due to the variation in both NO_3-N and NH_4-N throughout the study, the impact of these characteristics is not clear, but it does appear that NH_4-N in plot A2 is more like the SSSI, and ordination shows that this variable did have an influence on grouping similarities between Site A and SSSI quadrats. The compaction on Steart Marsh could have increased denitrification in soil, which converts nitrates into N_2O gas that is potentially harmful to the environment (Nawaz, Bourri and Troland, 2013). Future studies should make use of a flux chamber and mass spectrometry on these sites to assess which gasses are emitted from the surfaces of the different plots.

On both Site B plots in August 2017, PO_4-P levels were significantly higher than in the SSSI soil cores (C1 and C2). B2 C1 PO_4-P was also higher than in both Site A plots in 2017. Phosphate levels were originally high on the Site B plots in comparison to the Site A plots, and they remained similar to the starting state level until 2017 when they almost doubled. Areas on coastal wetlands that accumulate water can act as sinks for pollutants (Álvarez-rogel, Jiménez-cárceles and Nicolás, 2006), which may have been the case due to the submerged sediment on the plots. Additionally, the lack of vegetation on the site (see Chapter 4) may have meant there was less uptake of this nutrient, which saltmarsh plants use to aid their growth (Lillebø *et al.*, 2004). OM had lower levels of PO_4-P than the Site B plots each year, and vegetation coverage was also low on these sites, but OM had eroded by the end of the study, whereas the Site B plots had accreted. Therefore, the sediment that had arrived from the estuary may have been high in phosphorus, but the lack of plants rooted in the sediment may have left pools of the nutrient in the soil. Phosphorus does not move easily through sediment (Quintana, Moreno-amich and Comin, 1998), which explains why it did

not radically change in the first two years post-breach. It is likely that original PO_4-P would have been buried by the sediment that arrived on Steart Marsh, but erosion on OM may have mobilised phosphates in the initial stages post-breach, which can happen when terrestrial systems are breached (Weis and Butler, 2009); however, levels were significantly higher on OM by 2017. This macronutrient can be a pollutant, so high concentrations in the Site B sediment may be detrimental if these PO_4-P levels continue to rise.

When analysed with August 2014 soil data, the SSSI soil characteristic groupings were primarily governed by EC, % soil moisture and % SOC in C1 and C2 cores, which means that this site had higher levels of these variables in the soil. This was not unexpected, because saltmarshes are known to store SOC (e.g. Callaway *et al.*, 2012; Ouyang and Lee, 2013), and these habitats are regularly inundated by saline water. At this stage, ordination showed that % SOC was also having an influence of the grouping of 50% of the Site A quadrats in C1 cores, but these groupings were also governed by nitrate and ammonium. By August 2017, the Site A quadrats were grouped more closely to the SSSI soil cores at both 10cm and 10-20cm depths, because they had similar soil characteristics. This indicated that these sites had soil properties that could aid the development of similar plant communities to the SSSI. The soil data indicates that the Site B plots may not develop as quickly due to the potential soil compaction at lower levels and the standing water that remained for long periods. Areas like this are common on natural saltmarshes (salt pans) and have high levels of salinity that restrict plant growth (Adam, 2002), but these areas may be beneficial for wading birds if invertebrates colonise (Hughes, 2004). The colonisation of invertebrates on Steart Marsh and OM will be discussed in Chapter 5.

A feature that comes across clearly within these data is the decrease in variability between quadrats over time. This suggests that the quadrats within sites were becoming more similar in soil characteristics over the duration of the study. It is likely that accretion played a role in this change.

To summarise, it appeared that the Site A plots were more comparable to the SSSI in terms of soil variables and reached these similarities more rapidly than the other plots. However, this was not true of all quadrats, and some of the Site B quadrats did have similarities by 2017, although phosphate was higher in most of these quadrats. OM was different to all other plots through the study, and most soil variables were at the lowest levels in these plots. Accretion was rapid on Steart Marsh, but OM eroded during the study. Compaction and the effect of standing water on the Site B plots was a key parameter that was likely to affect vegetation succession on these plots, although it is important to note that

areas under water for prolonged periods of time are common in natural saltmarshes, and do have some benefits, especially if invertebrates colonise.

CHAPTER 4: VEGETATION SUCCESSION

4.1 Introduction

Vegetation is a key component of succession, and the establishment of halophytic plant communities is an important aim of saltmarsh restoration, especially because plants are a good indication of the performance of restored marshes (Garbutt and Wolters, 2008). Saltmarsh plants are important components of saltmarshes and provide grazing opportunities for fauna, including birds and invertebrates (Doody, 2008). They also help to provide ecosystem services such as flood protection, erosion control, water purification and nutrient cycling (Barbier, 2011).

The aim of this chapter is to report and summarise vegetation species composition on Steart Marsh in comparison to the SSSI saltmarsh (target state) and the managed OM site, and determine trends of community changes in relation to the target state throughout the study period using Detrended Correspondence Analysis, Cluster Analysis and National Vegetation Classification (NVC) analysis.

It was deemed important that pre-breach base-line data were collected of vegetation coverage on each study site to assess how quickly the MR changed from existing vegetation in relation to OM that still had agricultural plants and ruderal species. Collection of these data was especially important because the Steart Marsh and Otterhampton Marsh plots had different starting states, so a record of existing vegetation was key.

Species composition and NVC community matches for study plots are reported each year from 2014 (pre-breach) to August 2018. Detrended Correspondence Analyse is used to show how species composition in study plots (A1, A2, B1, and B2) relate to the target state and OM each year of the study. This provides information on the rate of vegetation colonisation and change in community composition in the different study plots, clearly showing which plots are changing at the fastest rates in terms of vegetation communities in comparison with the target state and how this relates to the managed coastal wetland site. In this chapter, cluster analysis shows similarities between quadrats across all plots, and the DCA ordinations highlight which species were responsible for similarities in quadrat positions. The results of these analyses are summarised in this chapter.

Previous studies have shown that colonisation of halophytic saltmarsh plants is rapid, but community composition that is comparable to mature marshes may take a considerable amount of time due to the prevalence of bare ground on MR sites (Mossman *et al.*, 2012;

Mossman, Davy and Grant, 2012; Brooks *et al.*, 2015). Therefore, it was expected that colonisation would be rapid on the Steart Marsh study plots because the study sites are close to a seed bank (SSSI), but it was likely that anaerobic conditions, unsuitable topography and compaction may limit the establishment of communities comparable to the SSSI during the course of the study.

4.2 Method

In each of the five permanent 2 x 2 metre quadrats vegetation was sampled twice a year (April and August) from August 2014 to August 2018, with plant community compositions recorded in the field as % cover values for all plant species in each quadrat. Community composition was classified using the National Vegetation Classification Scheme (NVC) after first converting the % cover values to Domin values (Table 3.1). Percentage cover data also facilitated the use of multivariate techniques to assess changes over time.

The SSSI reference site was assessed for vegetation coverage to ascertain a target state. This was carried out in August 2015 due to a lack of access in August 2014. The site was assessed again in 2016 to see if there was variation. It was determined that the variation was minimal, so the 2016 vegetation data was used as the target state.

In this study plant species were identified using the Colour Identification Guide to the Grasses, Sedges, Rushes and Ferns of the British Isles and North-Western Europe (Rose, 1989), The Wild Flower Key: How to Identify Wild Flowers, Trees and Shrubs in Britain and Ireland (Rose and O'Reilly, 2006), New Flora of the British Isles (Stace, 2010), and Plants and Habitats: An Introduction to Common Plants and Their Habitats in Britain and Ireland (Averis, 2013). Nigel Cox, a horticulturist, and Sarah Bolt, a member of the agricultural community also assisted with species identification. Saltmarsh plants were identified using the Guide to the Saltmarsh Plants of Britain Field Studies Council (FSC) guide (Oldham and Roberts, 1999), and on-line British plant databases (Ecoflora and Online Atlas of the British and Irish Flora).

4.2.1 Statistical analysis

Data were analysed using the NVC method utilising Match version 1.4 and MAVIS (Modular Analysis of Vegetation Information System) Plot Analyser version 1.04, which both provide a percentage match of the sampled community to NVC communities. NVC data were analysed on both programs to compare methods and gain more information to assess results in relation to British plant communities detailed by Rodwell (2000). Match has

been traditionally used for NVC analysis and uses Domin values to calculate community matches (Table 4.1), but a category for bare ground is not included on this program. MAVIS uses percentage coverage or Domin and takes bare ground into account if this category is inputted. It was anticipated that there would be substantial amounts of intertidal sediment covering the study sites after initial inundation, so MAVIS was used alongside the traditional method because of the inclusion of a bare ground category in the program. The top three community matches derived from both programs were compared.

Table 4.1 Domin scale used for NVC analysis on the Match program.

% coverage	Domin value
91-100	10
76-90	9
51-75	8
34-50	7
26-33	6
11-25	5
4-10	4
<4 (many)	3
<4 (several)	2
<4 (few)	1

Cluster analysis, using Euclidean distance and the average linkage method, was conducted to assess similarities between quadrats on all sites, and ordination was used to show which species were responsible for groupings of quadrats. These analyses were carried out on all quadrats each year (2014-2017). Principal Component Analysis (PCA) was initially chosen as an ordination method, but biplots showed that there was a horseshoe effect. To rectify this, Detrended Correspondence Analysis (DCA) was conducted. This method removes the horseshoe effect that commonly occurs in PCA and the arch effect that is associated with Correspondence Analysis. It does this by dividing axis 1 into sections and centring axis 2 on zero. Although this method has the limitation of reducing the relevance of eigenvalues compared to the other forms of eigenanalysis, it has the benefit of generating biplots that are easier to interpret and displays a more realistic representation of data. It was decided that eigenvalues ≥ 0.5 on an axis would be deemed as a meaningful environmental gradient in the DCA analysis. DCA ordination was carried out on R version 3.3.2 using the Vegan package (Oksanen *et al.*, 2017). Cluster analysis was performed on Minitab version 18.1. Descriptive data were created on Minitab version 18.1.

4.3 Results

4.3.1 Pre-breach vegetation coverage

Throughout the study period (August 2014-2018) 44 plant species were identified. Thirty-three of these species are commonly found on terrestrial habitats and 11 are characteristic of saltmarshes. Additionally, algal mat, which is characteristic of saltmarshes, was found during the study, and unidentified green algae were also detected (Table 4.2). The dominant vegetation coverage on each plot on Steart Marsh (A1, A2, B1, and B2) and OM pre-breach in August 2014, and on the SSSI are shown in Figure 4.1.

In August 2014 Site A resembled a grassland meadow, which consisted of grass species such as *Agrostis stolonifera and Holcus lanatus,* which had high coverage on both plots, with ruderal species, including *Cirsium arvense* and *Cirsium vulgare,* being recorded. *Lolium perenne* was also detected amongst the other grasses, along with common bent (*Agrostis capillaris*). Meadow barley (*Hordeum secalinum*) was also found in the quadrats along with a low coverage of other flowering plants, including white clover (*Trifolium repens*) on A1 and broad-leaved willowherb (*Epilobium montanum*) on both plots. There was very little bare ground on Site A before the breach (Mean = 2% A1, and 0.8% A2). In total, fourteen plant species were found on A1, and fourteen were found on A2 in August 2014.

Seventeen plant species were found on the B1 plot in August 2014 and twenty-two were found on B2. There was a wider range of species compared to Site A plots due to the disturbed habitat, but percentage coverage of these species was relatively low and bare ground was higher on these plots. Three species of dock were found on the Site B plots and there was more white clover. Bristly oxtongue (*Picris echioides*) was found in three out of five quadrats on B1 and on B2. The majority of grasses found on the Site B plots were the same species that were detected on the Site A plots, including *A. stolonifera and H. lanatus*, but their coverage was generally sparser, which was expected on a site that was previously in an arable crop rotation. The only obvious remnants of crop found on the plots was rapeseed (*Brassica napus*), but coverage was very low and only found in two out of five quadrats on B1. However, more of this crop was seen between and around the plots during site preparation and during data collection in August 2014.

Table 4.2 Plant species found in quadrats during the study period (August 2014-2018). Species with a green shaded background are terrestrial and species with a blue shaded background are characteristic of saltmarshes. Blue-green algae (shaded light brown) was not identified to species. * Species which can also be found on saltmarshes.

Species	Common name
Agrostis capillaris	Common bent
Agrostis stolonifera	Creeping bent*
Alopecurus pratensis	Meadow foxtail
Apium graveolens	Wild celery
Avena fatua	Common wild oat
Brachythecium rutabulum	Rough-stalked feather moss
Brassica napus	Rapeseed
Bromus sterilis	Barren brome
Cirsium arvense	Creeping thistle
Cirsium vulgare	Spear thistle
Cynosurus cristatus	Crested dog's-tail
Dactylis glomerata	Cock's foot
Elymus repens	Common couch
Epilobium montanum	Broad-leaved willowherb
Holcus lanatus	Yorkshire fog
Holcus mollis	Creeping soft grass
Hordeum secalinum	Meadow barley
Juncus effusus	Soft rush
Lolium multiflorum	Italian ryegrass
Lolium perenne	Perennial ryegrass
Matricaria perforata	Scentless mayweed
Medicago lupulina	Black medic
Phleum bertolonii	Smaller catstail
Phleum pratense	Timothy-grass
Picris echioides	Bristly oxtongue
Plantago major	Greater plantain
Poa annua	Annual meadow grass
Pulicaria dysenterica	Common fleabane
Ranunculus parviflorus	Small-flowered buttercup
Rumex crispus	Curled dock
Rumex obtusifolius	Broad-leaved dock
Rumex sanguineus	Red-veined dock
Trifolium repens	White clover
Atriplex portulacoides	Sea purslane
Atriplex prostrata	Spear-leaved orache
Aster tripolium	Sea aster
Cochlearia anglica	English scurvy-grass
Elytrigia atherica	Sea couch
Puccinellia maritima	Common saltmarsh grass
Salicornia europaea	Glasswort
Spartina anglica	Common cordgrass
Spergularia marina	Lesser sea-spurrey
Spurgularia media	Greater sea-spurrey
Suaeda maritima	Annual seablite
Algal mat	N/A
Green algae spp.	N/A

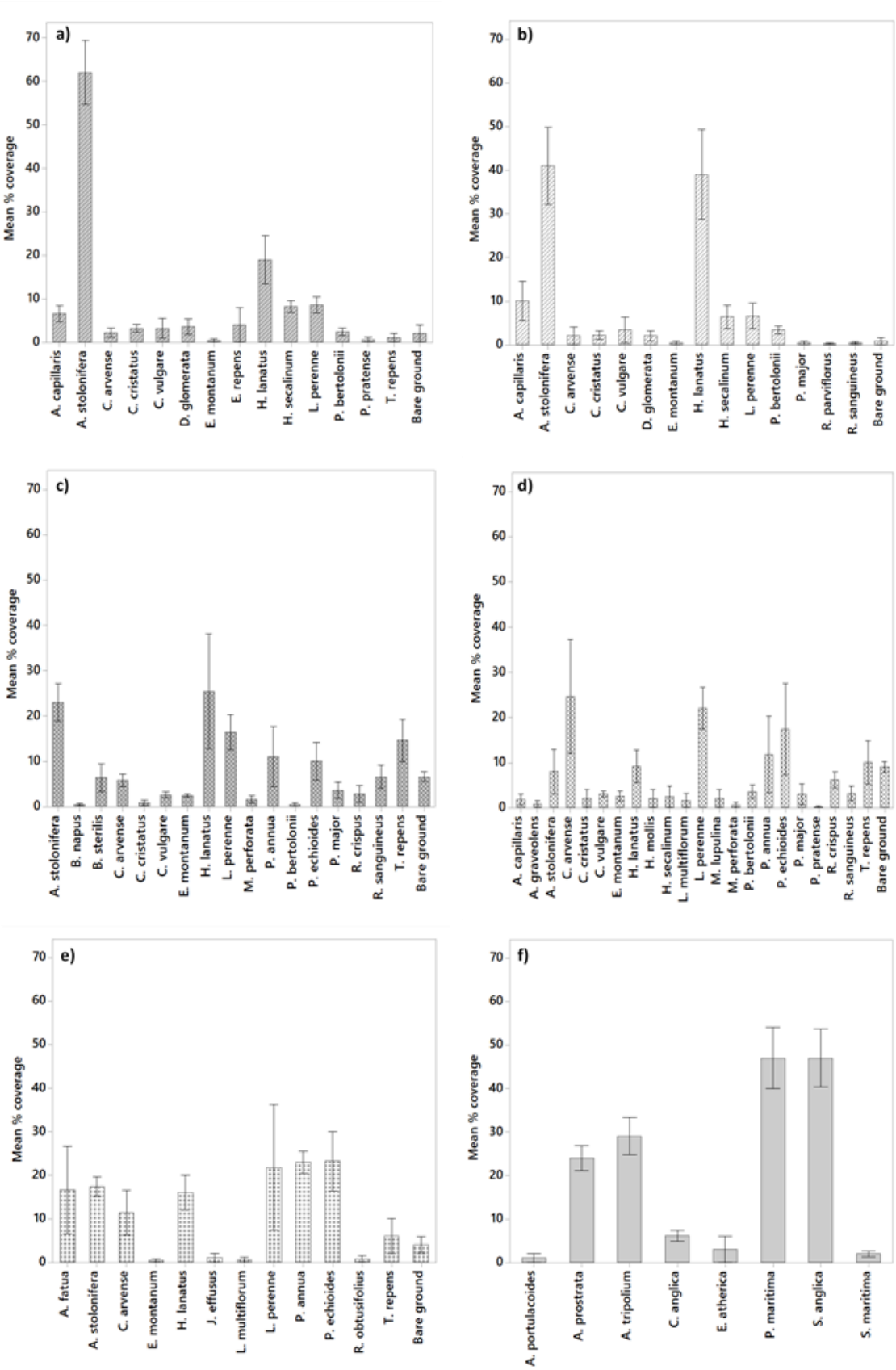


Figure 4.1 Mean vegetation coverage on sample plots a) A1, b) A2, c) B1, d) B2, e) OM and f) SSSI in August 2014 (SSSI data is from August 2016). Figures on y axes represent average percentage coverage of species in $2m^2$ quadrats (n = 5) on each site. Error bars represent standard error of the mean.

Twelve plant species were found on OM in August 2014. The site was generally like the Site B plots, but species richness was lower and there was less bare ground. The same grasses were present on the site, and ruderal species were detected in the quadrats. *Juncus* was found in OMQ4, but not found in the other quadrats. Common wild oat (*Avena fatua*) was seen on OM, but not on any of the other study sites.

The SSSI (recorded August 2016) was fully vegetated with a dense sward. Diversity was low as was expected on a saltmarsh, but some characteristic species were not present, such as sea lavender (*Limonium vulgare*) and *Salicornia* spp. The dominant species were the grasses *Spartina anglica* and *Puccinellia maritima*, but sea aster (*Aster tripolium*) and spear-leaved orache (*Atriplex prostrata*) were also relatively abundant. *Suaeda maritima, Elytrigia atherica, Cochlearia anglica,* and *Atriplex portulacoides* were found on the SSSI plot, but coverage was low. *E. atherica* was only found in one quadrat (SSSI Q4) and *A. portulacoides* was only found on the outer edge of SSSI Q1 (Figure 4.2 and 4.3).

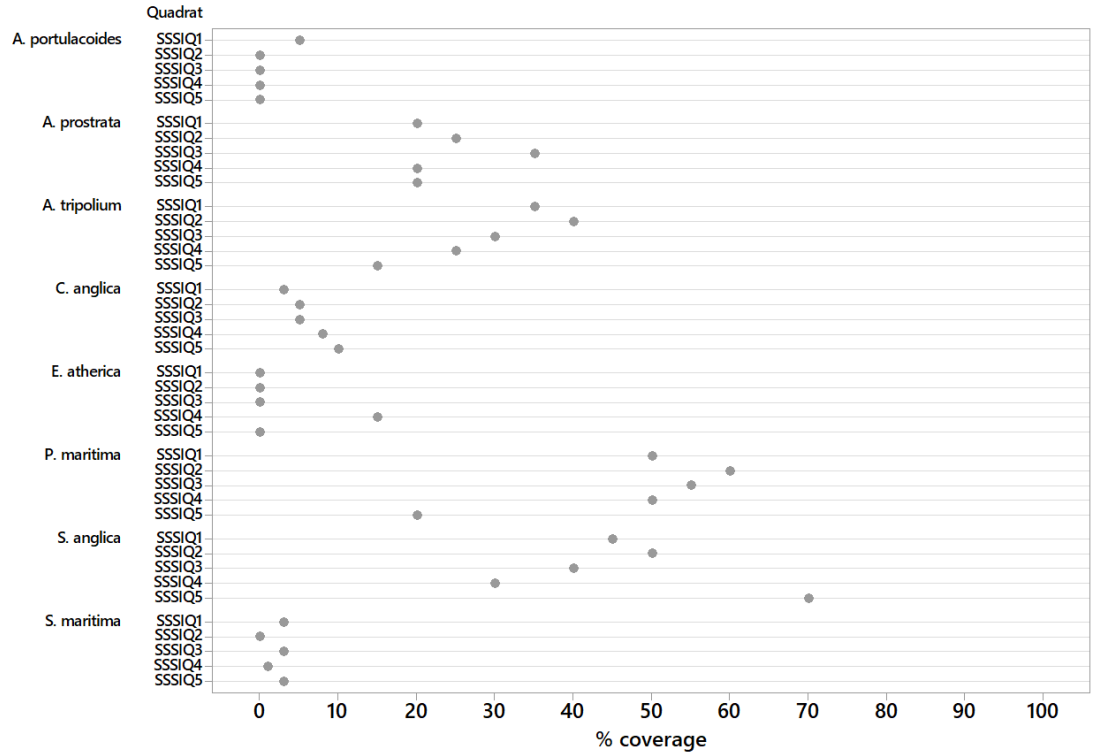


Figure 4.2 Vegetation coverage in each SSSI quadrat in August 2016.

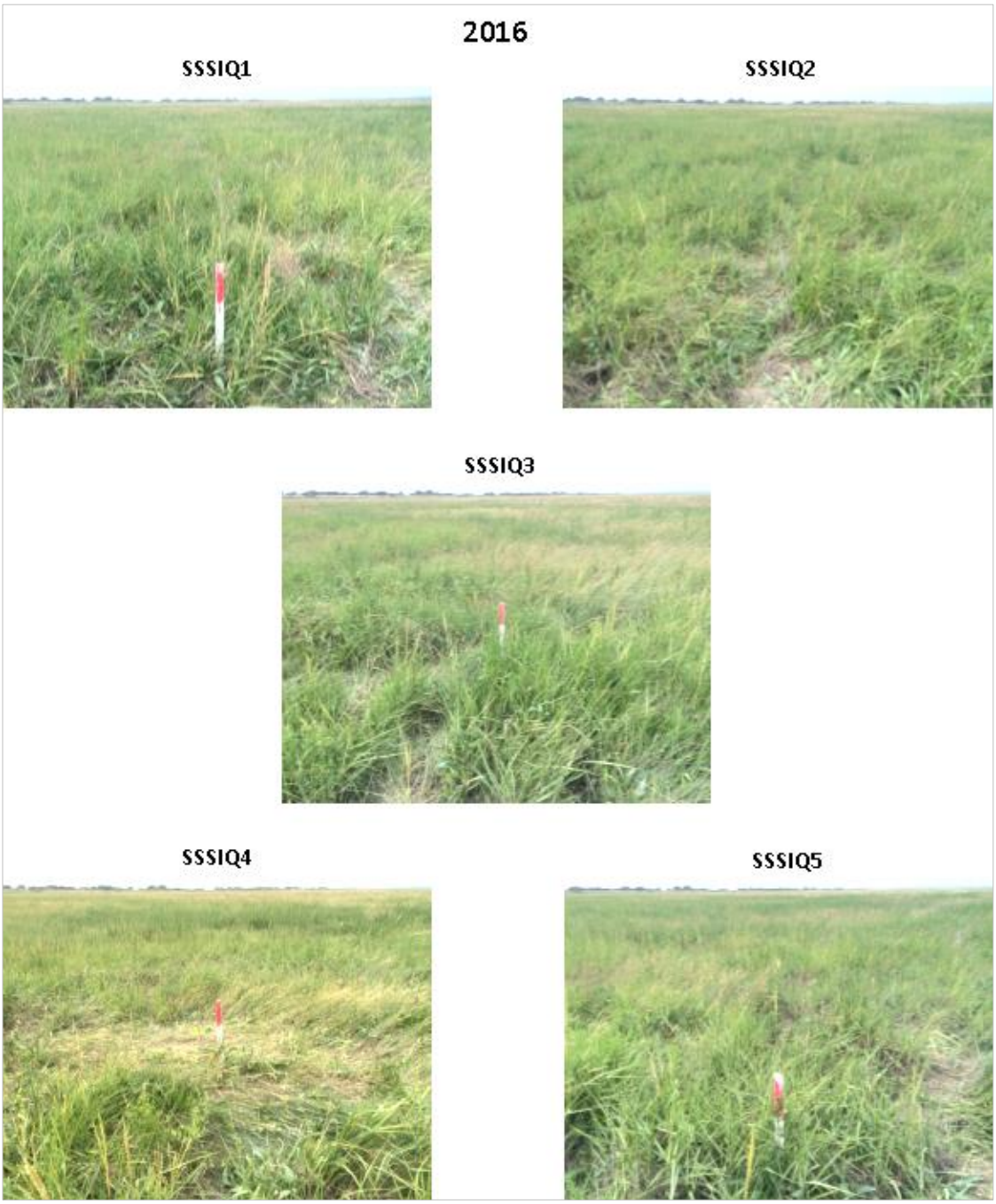


Figure 4.3 Photographs of vegetation in each quadrat on the SSSI in August 2016.

4.3.2 Post-breach vegetation coverage

4.3.2.1 Site A vegetation succession

When the permanent quadrats were observed again in April 2015 (post breaching) it was discovered that bare ground had risen to 78% in A1, and 83% in A2. *A. stolonifera* was still present on both sample areas, although there was less coverage, and *H. lanatus* had disappeared from A1, and was reduced in A2. In April, a small percentage coverage of saltmarsh species, including *A. prostrata* and *C. anglica* were identified in the quadrats, while many areas were starting to form extensive mudflats (Figure 4.4). This, along with

dead terrestrial vegetation in the quadrats, indicated the beginning of a period of change following the implementation of the tidal regime.

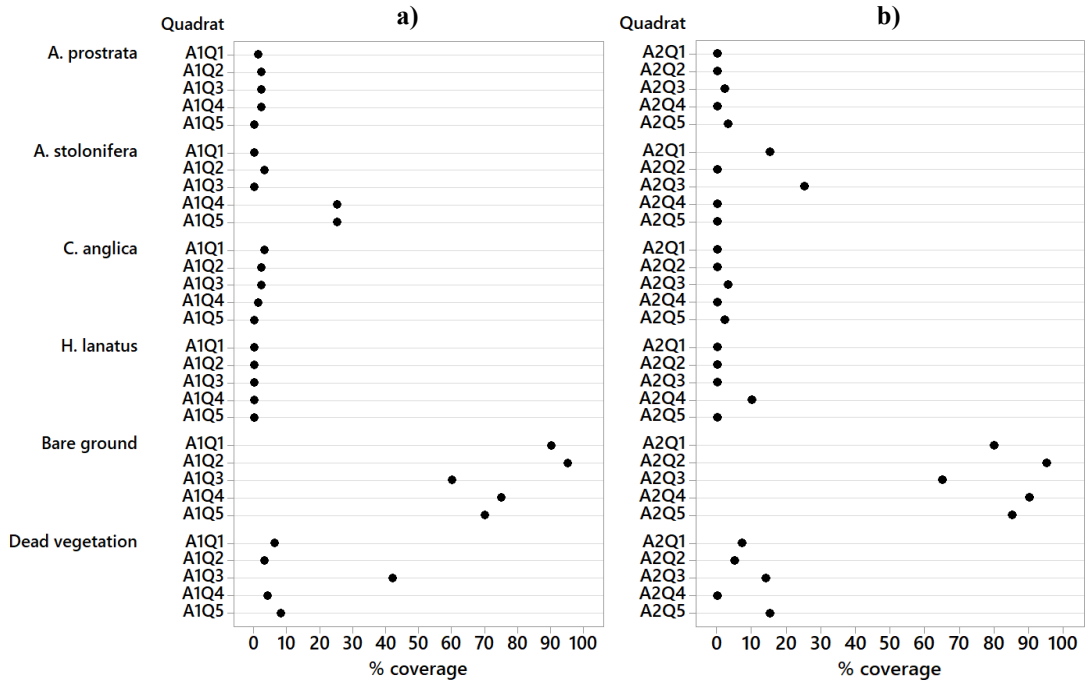


Figure 4.4 Vegetation coverage in each quadrat in a) A1 and b) A2 in April 2015.

In August 2015 *A. prostrata* coverage was substantially higher in the Site A plots than it had been in April, chiefly due to further colonisation and growth of the plants that had seeded post-breach. *S. maritima* had begun to colonise the site, but bare ground was still relatively high in comparison to the vegetation coverage (Figure 4.5).

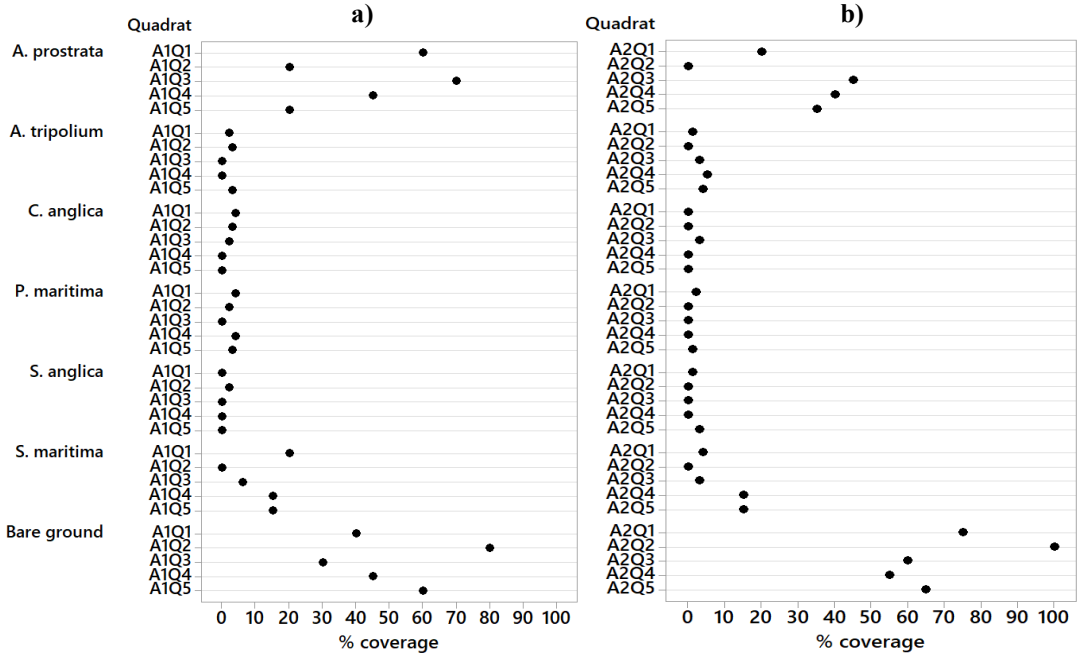


Figure 4.5 Vegetation coverage in each quadrat in a) A1 and b) A2 in August 2015.

Stands of *A. tripolium* were found on the Site A plots and other characteristic saltmarsh species had colonised in some quadrats, including the cordgrass, *S. anglica* and the saltmarsh grass, *P. maritima*. There were also small patches of the flower *C. anglica* in some of the quadrats on A1 and A2, and all four species had colonised further by April 2016. *A. prostrata* was still prevalent in April 2016, especially in A1Q3, but *S. maritima* was not found. There were still large patches of bare ground in all quadrats, but especially in A2Q1 and A2Q2 (Figure 4.6).

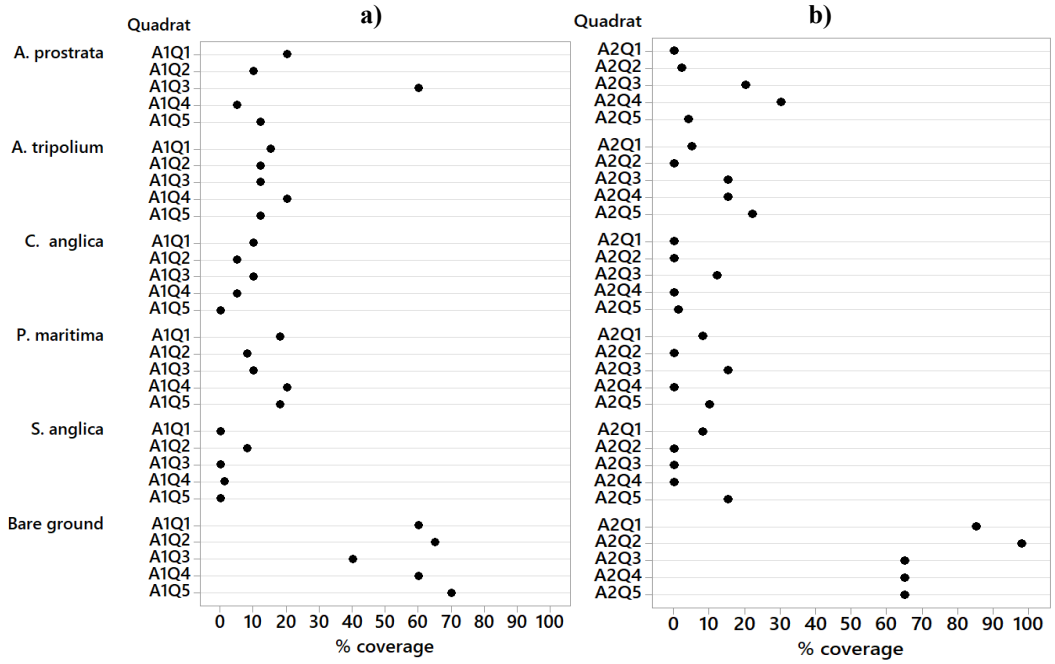


Figure 4.6 Vegetation coverage in each quadrat in a) A1 and b) A2 in April 2016.

An algal mat had started to form in A1Q1, A1Q4 and A2Q2 by August 2016, although this was minimal. *A. prostrata* was still prolific in most of the Site A quadrats, but *C. anglica* was no longer found. *A. tripolium* and *P. maritima* were becoming more abundant in most quadrats, and *S. anglica* was more prevalent than the previous year. Bare ground was still high in some quadrats, particularly A2Q1 and A2Q2, which both had > 50% bare sediment (Figure 4.7).

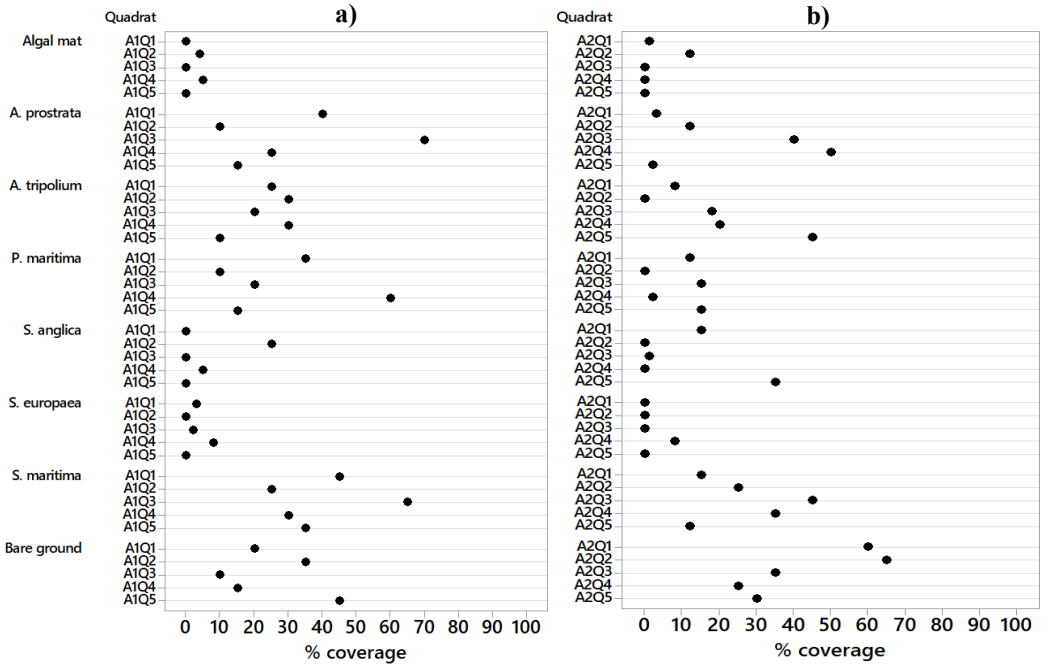


Figure 4.7 Vegetation coverage in each quadrat in a) A1 and b) A2 in August 2016.

By April 2017, *P. maritima* was growing well on Site A plots and was present in all quadrats. There were substantial stands of the species in some quadrats, especially on A1. Some of the large stands of *A. tripolium* had died back, but there was new growth coming through. Bare ground was still relatively high on A2, but this was less on A1. *A. prostrata* and *S. maritima* were not as prevalent as they had been during the initial stages post-breach (Figure 4.8).

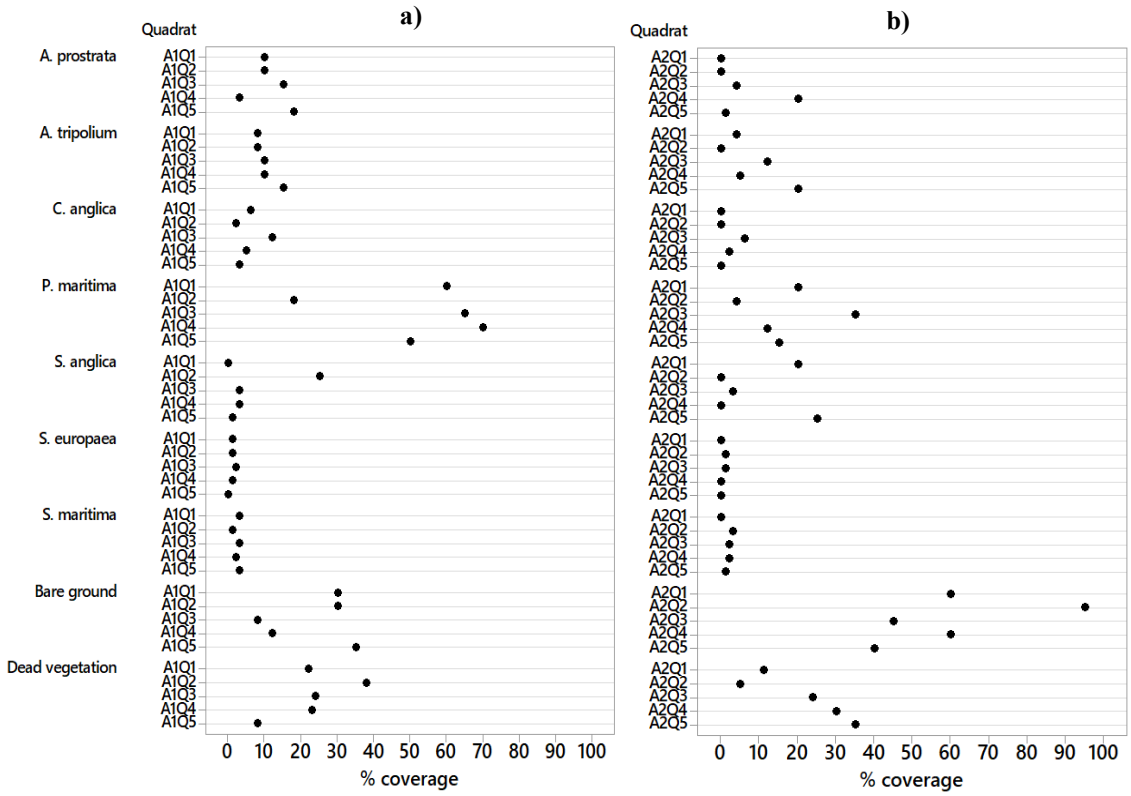


Figure 4.8 Vegetation coverage in each quadrat in a) A1 and b) A2 in April 2017.

By August 2017, most Site A quadrats had similar species composition to the SSSI, and bare ground was minimal on A1. There was more bare ground on A2, particularly in quadrat A2Q2, but there were large stands of *S. anglica* in A2Q2 and A2Q5. There were also large stands of this species in A1Q2. *P. maritima* had more coverage in the A1 quadrats, and *A. tripolium* had grown back and formed large stands. *S. media* (greater sea-spurrey) was found in one quadrat on A1, but coverage was very minimal (Figure 4.9).

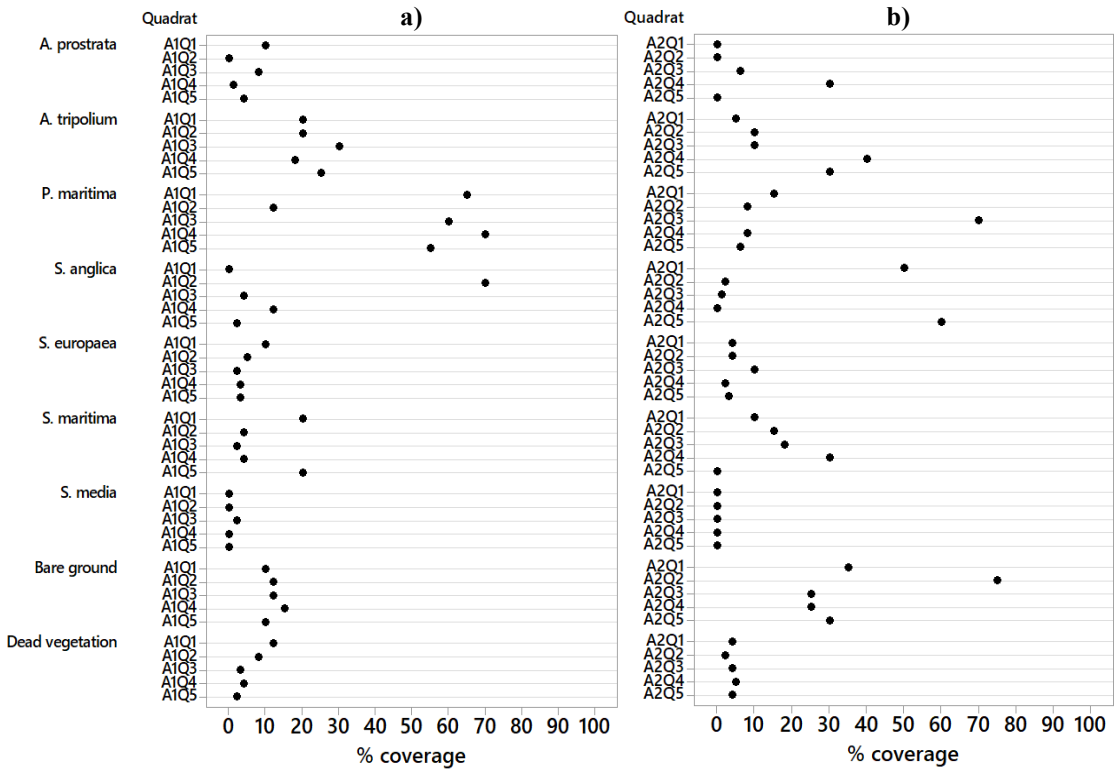


Figure 4.9 Vegetation coverage in each quadrat in a) A1 and b) A2 in August 2017.

4.3.2.2 Site B vegetation succession

It was apparent that the Site B plots were more compacted than the other sites (see Chapter 3), and held some water after the initial breaching, though this had dried out somewhat by April 2015. The April composition differed to that in August 2014 with a substantial extent of bare ground, and a reduction of other species. There was not any visible colonisation by halophyte plants on this site (Figure 4.10).

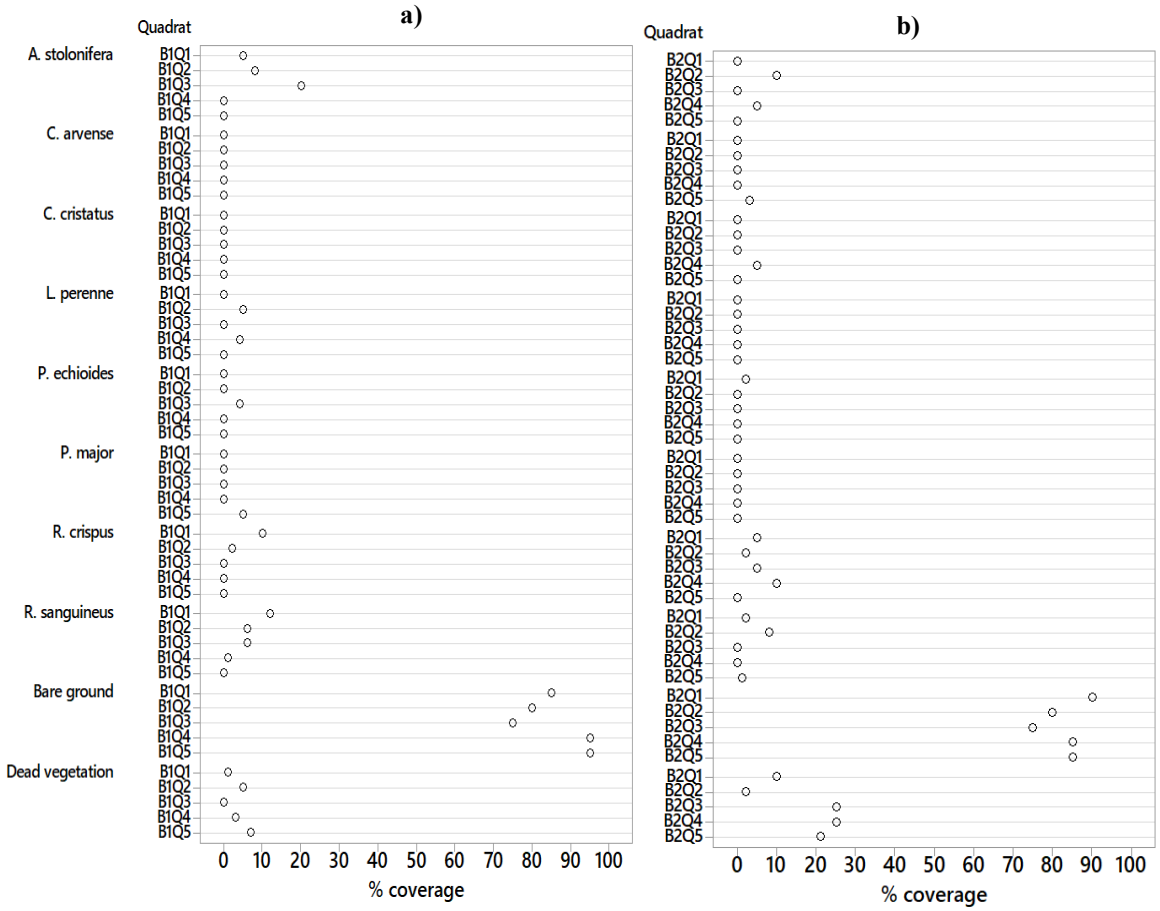


Figure 4.10 Vegetation coverage in each quadrat in a) B1 and b) B2 in April 2015.

By August 2015 some characteristic saltmarsh species had started to colonise on the Site B plots, but the coverage was very low, and the plants appeared in small clumps where present. The plots had been periodically under standing water, which impeded the growth of species, possibly due to anaerobic conditions, hence bare ground persisted (Figure 4.11).

Some halophytes, including *A. prostrata*, *P. maritima*, and *S. anglica* had colonised the Site B plots in August 2015, but coverage was very sparse (Figure 4.11).

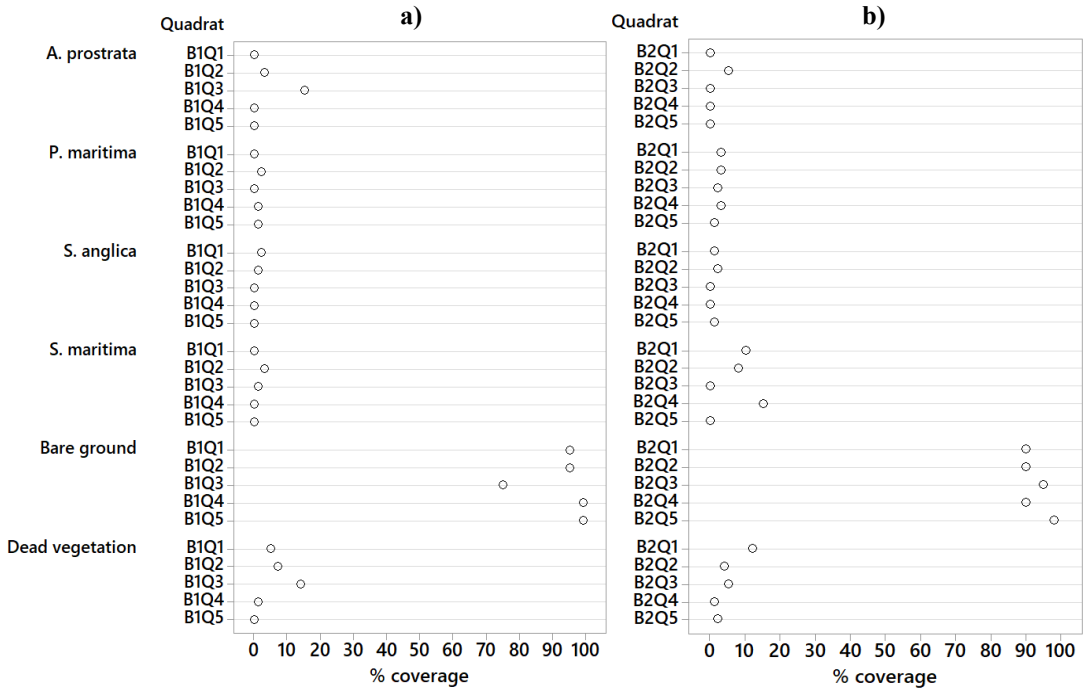


Figure 4.11 Vegetation coverage in each quadrat in a) B1 and b) B2 in August 2015.

Bare ground was persisting in these plots due to an aquiclude that had formed, which prevents the flow of water through the soil. Most of the dead vegetation had gone or was buried in sediment, but *P. maritima* coverage did start to increase by April 2016 (Figure 4.12).

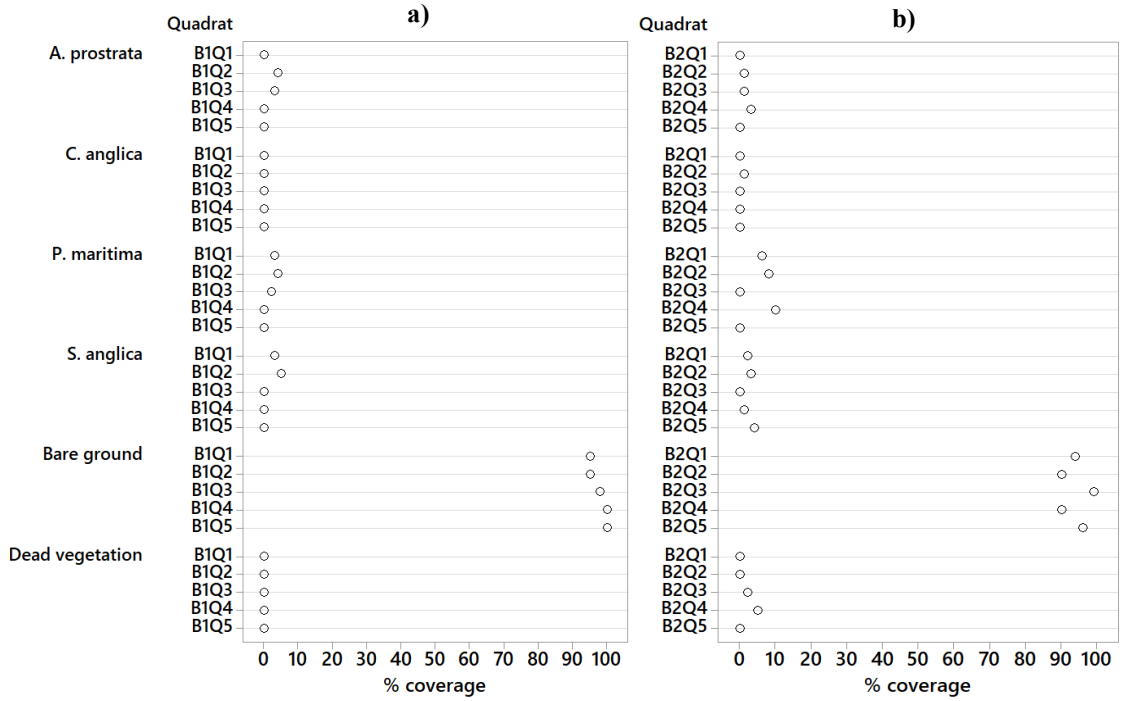


Figure 4.12 Vegetation coverage in each quadrat in a) B1 and b) B2 in April 2016.

Bare ground was still very extensive on the Site B plots in August 2016, and species richness was lower than on the Site A plots (5 species compared to 7 on Site A plots).

Nevertheless, there was further growth of some characteristic species on both B1 and B2 plots, including *P. maritima*, which was more abundant on B2. *S. maritima* was relatively high in some quadrats, and there were small patches of *S. anglica* that had grown since the previous year. *A. prostrata* had grown again on the site, but the most notable change was the colonisation by *S. europaea*, with numerous individuals established in some quadrats (Figure 4.13).

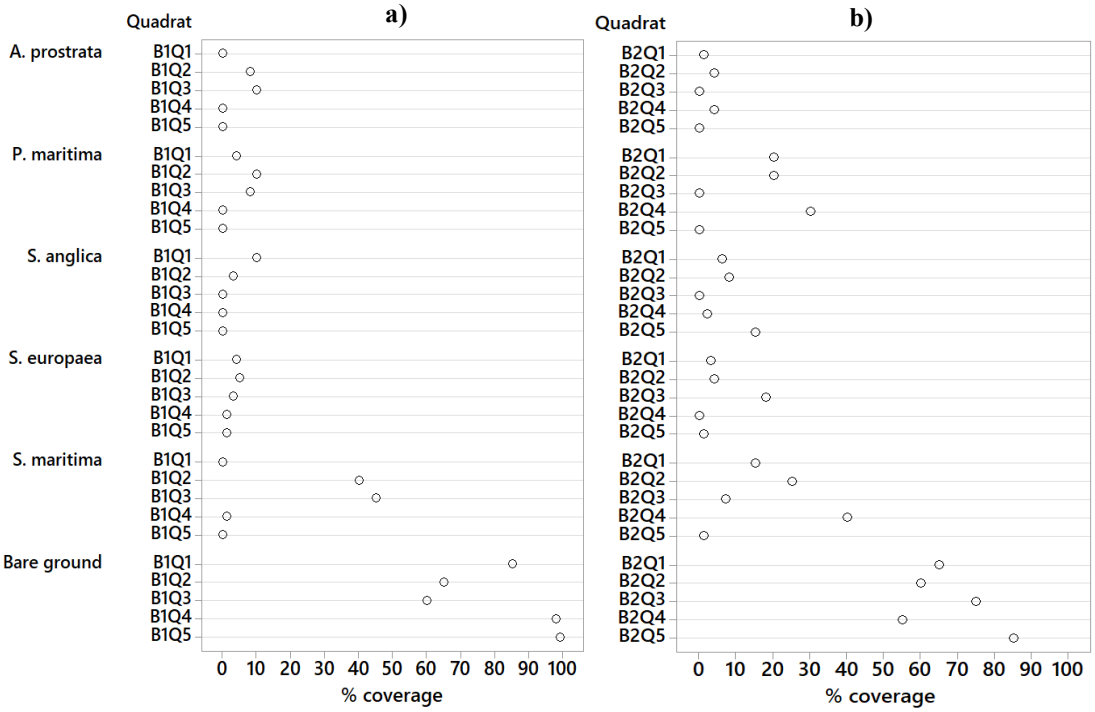


Figure 4.13 Vegetation coverage in each quadrat in a) B1 and b) B2 in August 2016.

P. maritima was growing more substantially on the Site B plots by April 2017, and *S. anglica* was more prevalent than it had been previously. *S. europaea* coverage had not changed markedly from August 2016, but *A. prostrata* was no longer seen in the quadrats. *A. tripolium* had colonised on B2, but there were only a few individuals. Bare ground was still substantial on the Site B plots in April (Figure 4.14).

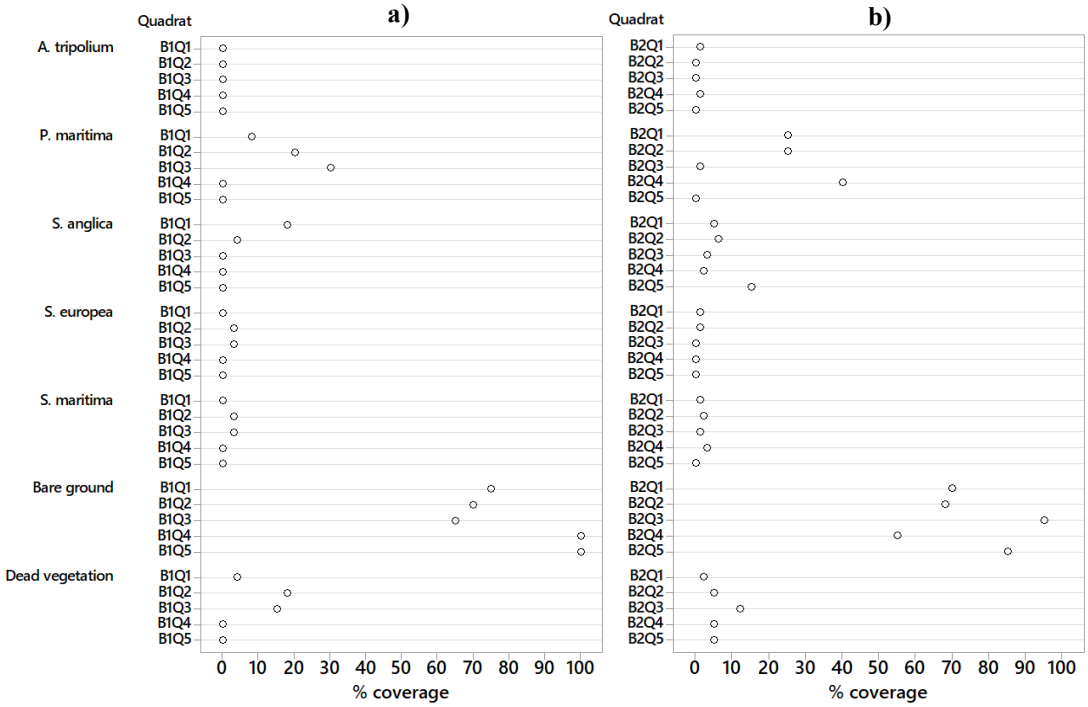


Figure 4.14 Vegetation coverage in each quadrat in a) B1 and b) B2 in April 2017

Some of the Site B quadrats still had 100% bare ground by August 2017, but other quadrats did have coverage of *P. maritima* that had been gradually growing since initial establishment. *S. europaea* was more prevalent than it had been previously in some quadrats. Stands of *S. anglica* had grown into substantial clumps in some B1 and B2 quadrats by August 2017 and measured up to 70cm in height. *A. tripolium* was very minimal and was only found in B2Q4 (Figure 4.15).

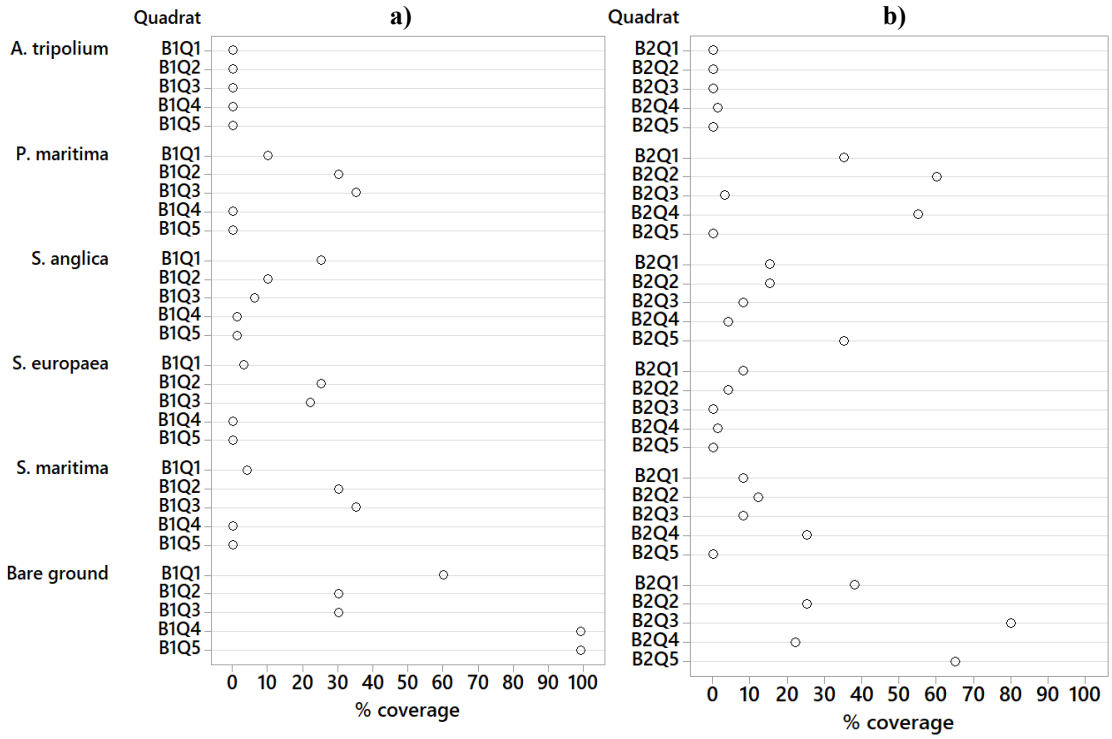


Figure 4.15 Vegetation coverage in each quadrat in a) B1 and b) B2 in August 2017.

4.3.2.3 OM vegetation succession

OM had not changed dramatically since August 2014, and similar plant species were present. The only species that were no longer detected in the quadrats were *R. obtusifolius* and *E. montanum* (Figure 4.16).

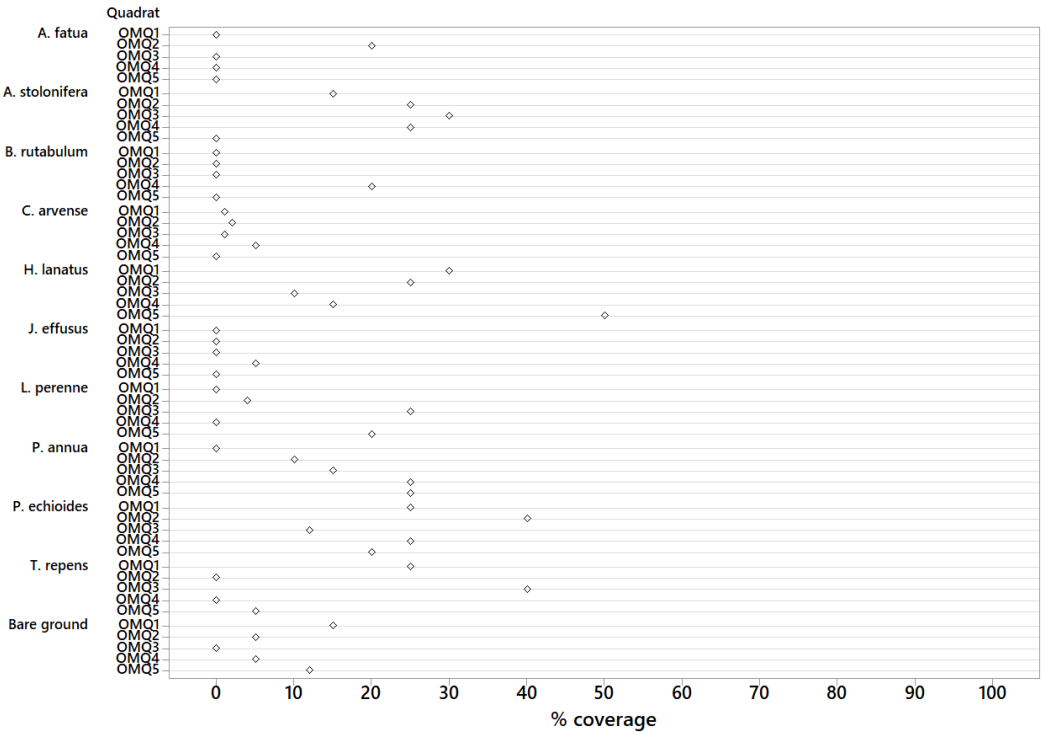


Figure 4.16 Vegetation coverage in each OM quadrat in April 2015.

Vegetation coverage did not change substantially on OM by August 2015, and characteristic saltmarsh species were yet to colonise this site. Terrestrial species were still present at this stage, but there was more bare ground and some existing species had died back (Figure 4.17).

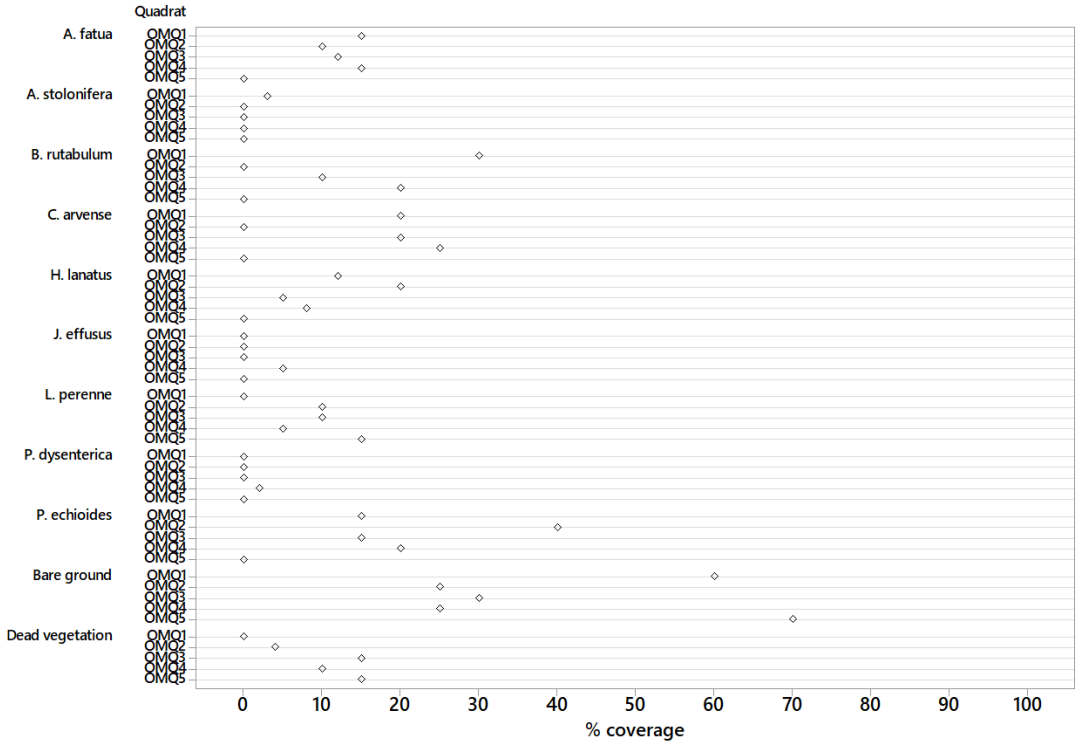


Figure 4.17 Vegetation coverage in each OM quadrat in August 2015.

Most of the vegetation had gone from OM by April 2016, but there were remnants of *L. perenne*, which was either from the previous year or it had colonised from nearby fields. *J. effusus* was still found in OMQ4. There was a lot of bare ground on the plot and there were numerous patches of green algae spp. in all quadrats, but especially in OMQ3 (Figure 4.18).

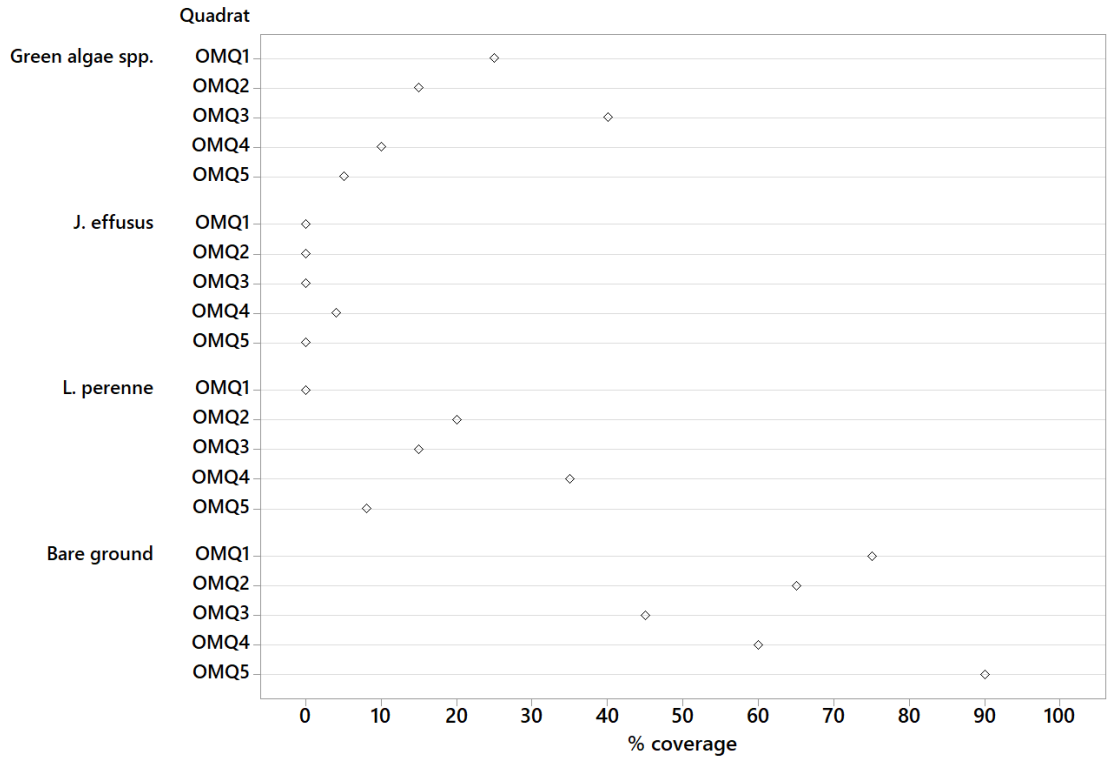


Figure 4.18 Vegetation coverage in OM quadrats in April 2016.

Bare ground was still extensive on OM in August 2016. Terrestrial species were observed, and algae were still detected, although only in OMQ1. Red fescue (*F. rubra*), which is sometimes found in the upper marsh of saltmarshes, was found in OMQ4, but this was minimal and could have colonised from nearby agricultural fields. *S. marina*, a *Spergularia* species commonly found on saltmarshes, was found in OMQ4, but coverage was very minimal (Figure 4.19).

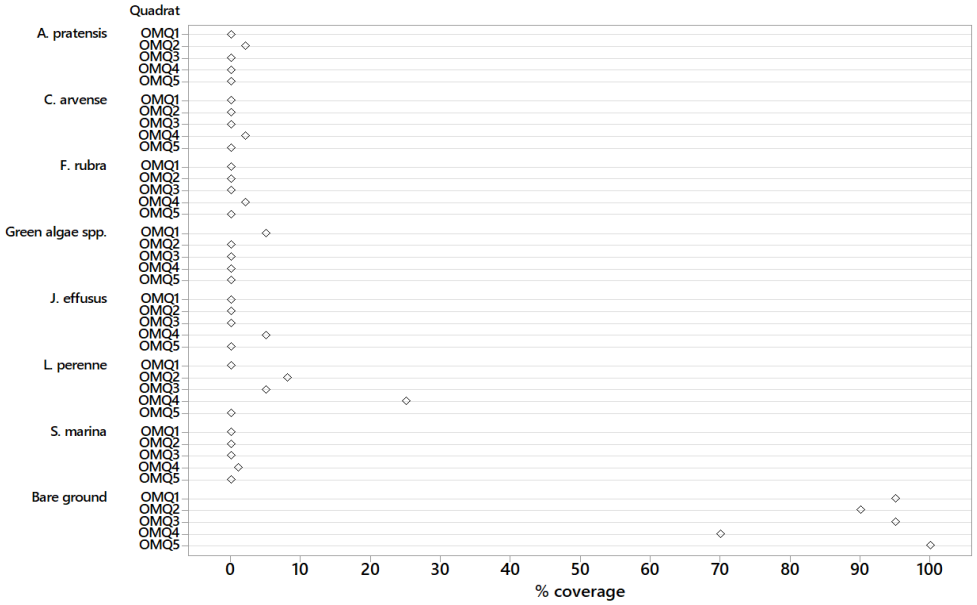


Figure 4.19 Vegetation coverage in OM quadrats in August 2016.

OM was not assessed in April 2017. Access was denied due to birds breeding on the site. It was part of the remit of this research that there would not be an impact on breeding birds during the study, and permission to access sites each year had to be gained before sampling could commence.

When the site was visited in August 2017 it was observed that there was still a large coverage of bare ground on the OM plot, but algae spp. were no longer seen. There were still some remnants of terrestrial species, but most of the growth on the plot was *Spergularia* species. *S. marina* was the most abundant species observed, although *S. media* was also recorded (Figure 4.20).

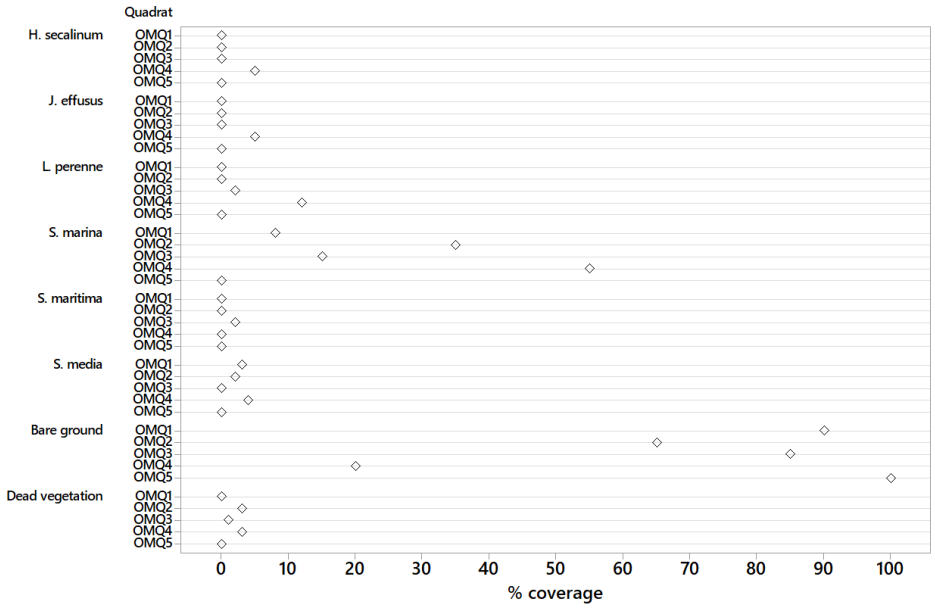


Figure 4.20 Vegetation coverage in OM quadrats in August 2017.

4.3.2.4 Photographic evidence of vegetation succession

Photographs were taken in late summer/early autumn each year to visually document the change of vegetation in each quadrat each year (Figure 4.21-Figure 4.25). This was carried out at the end of each sampling period and provides further evidence of successional changes at the height of the growing season. This time frame was chosen because it was appropriate to gather these qualitative data before annual vegetation died back. The prevalence of bare ground is clearly shown.

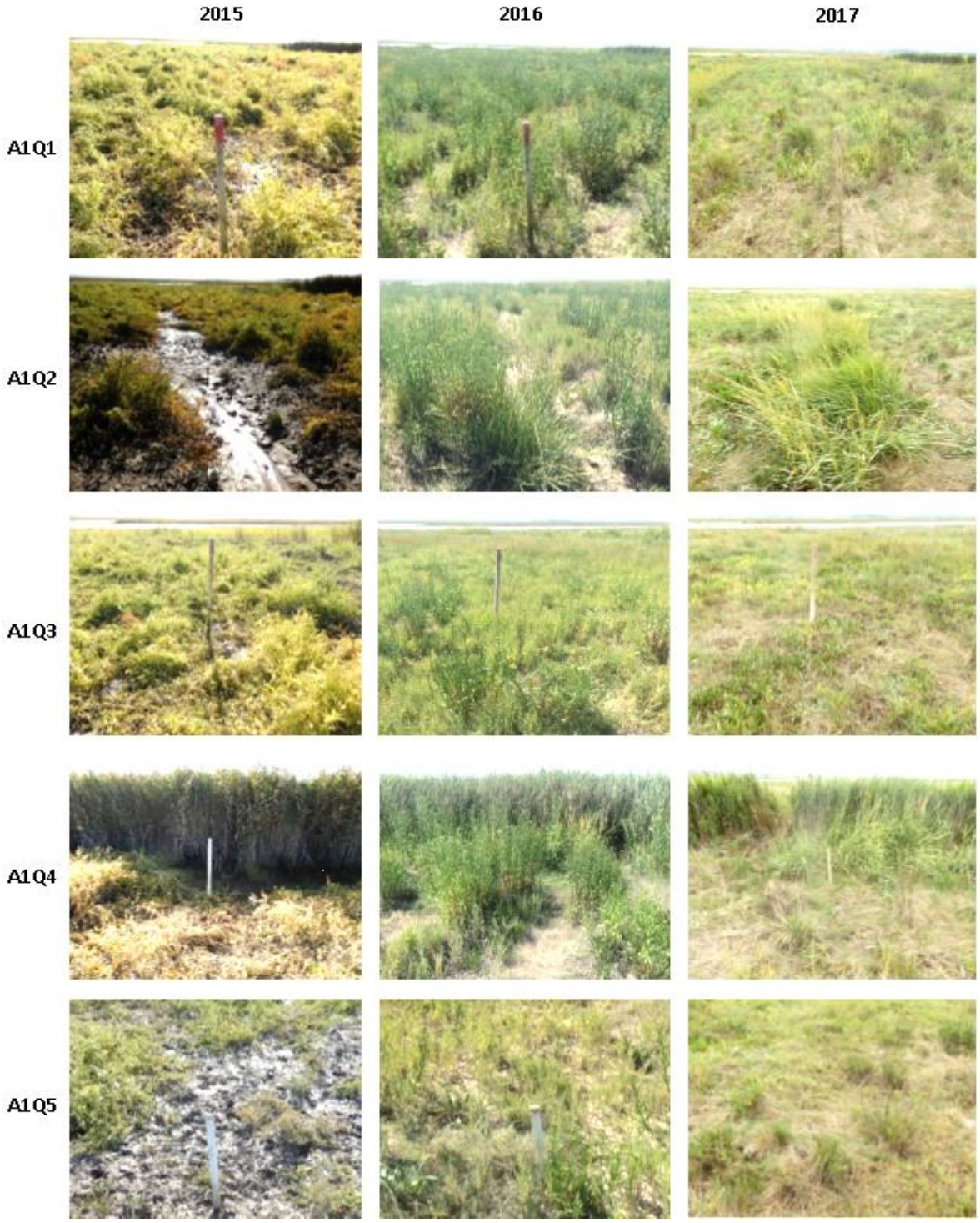


Figure 4.21 Photographs of plant succession on A1 quadrats in 2015-2017. Photographs were taken at the end of the late summer sampling periods each year. 2015 photos were taken in September of that year. 2016 and 2017 photos were taken in August of each year. Posts are missing from some 2017 photographs because they were broken by cattle. Q2 plots were not marked by posts.

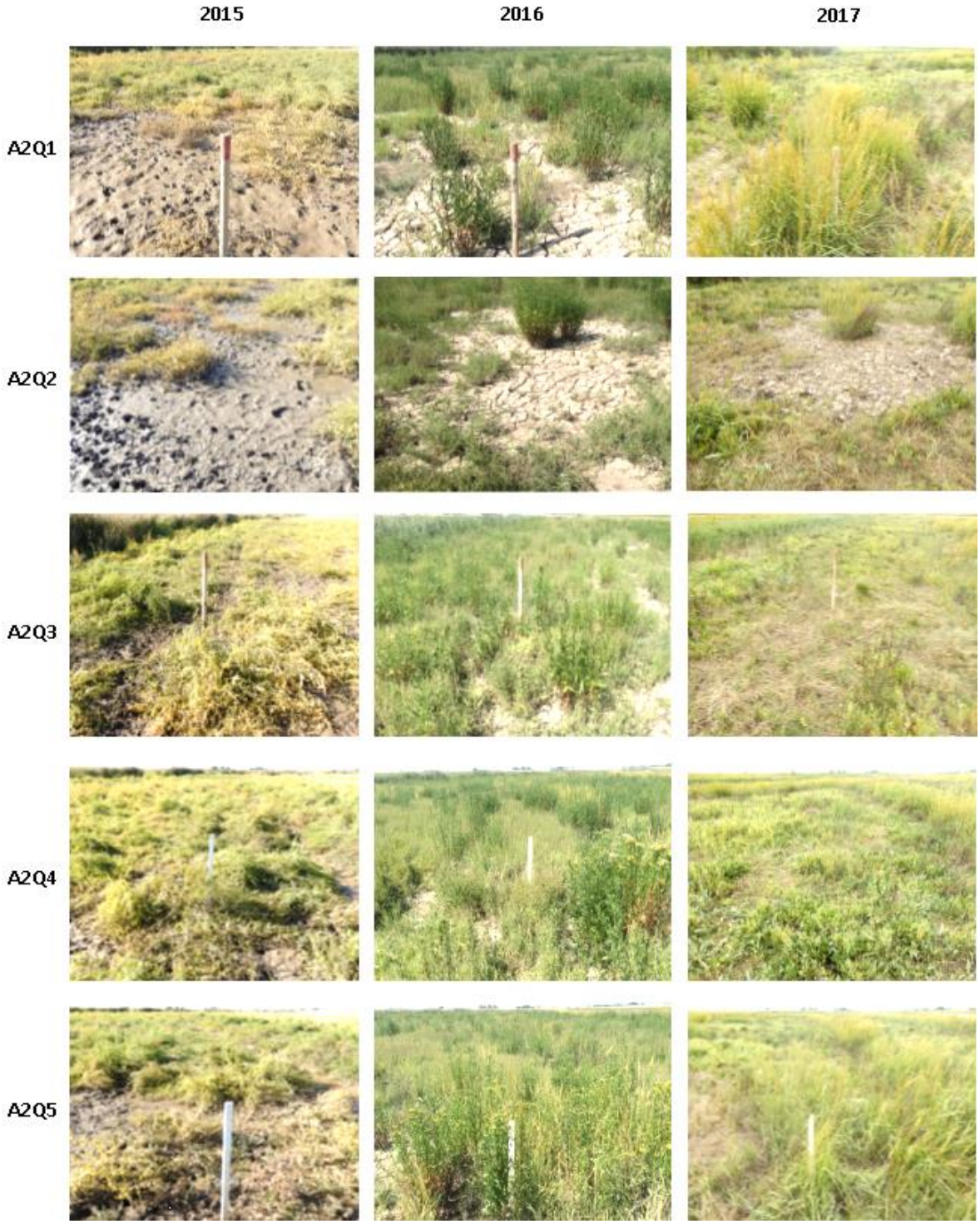


Figure 4.22 Photographs of plant succession on A2 quadrats in 2015-2017. Photographs were taken at the end of the late summer sampling periods each year. 2015 photos were taken in September of that year. 2016 and 2017 photos were taken in August of each year. Posts are missing from some 2017 photographs because they were broken by cattle. Q2 plots were not marked by posts.

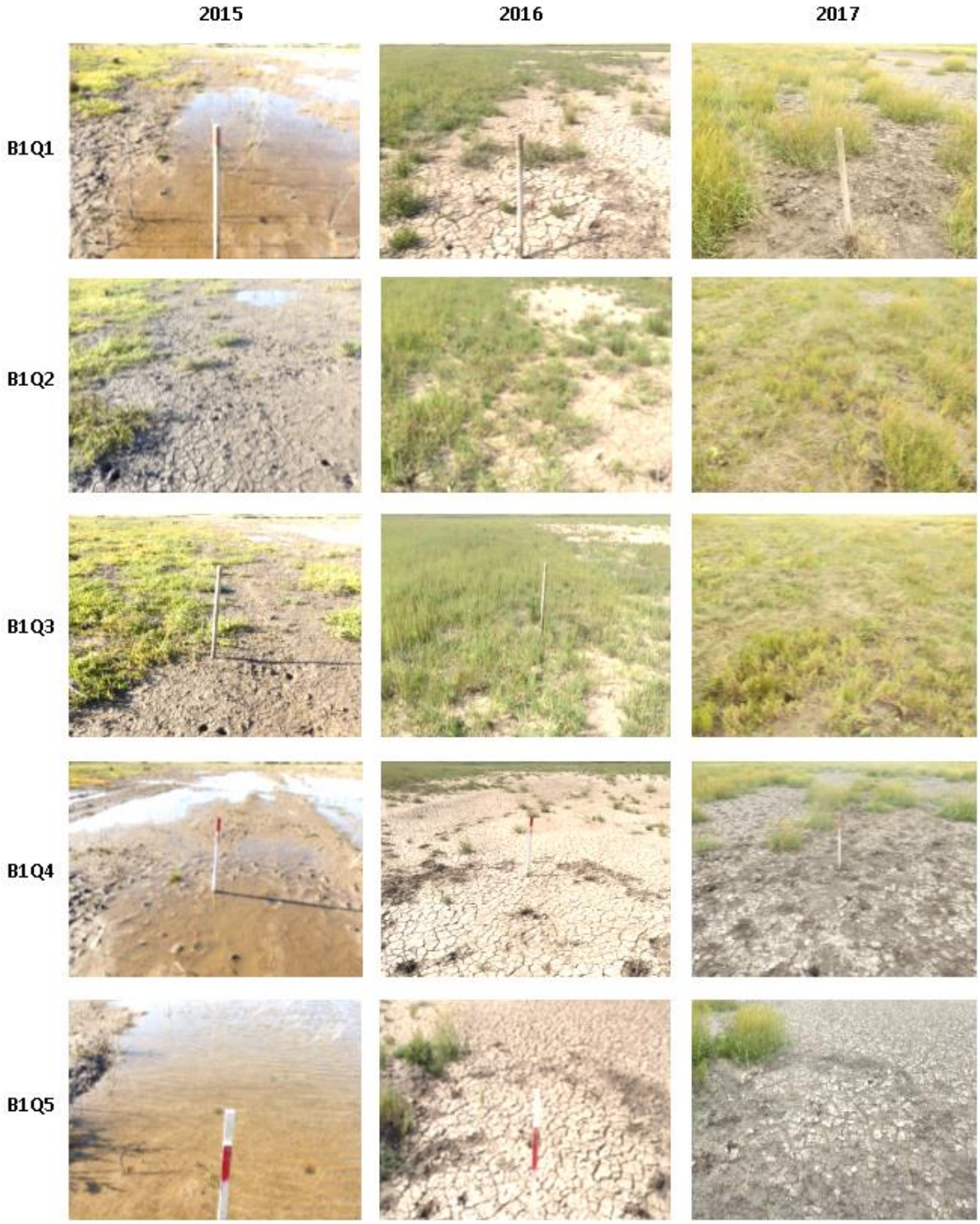


Figure 4.23 Photographs of plant succession on B1 quadrats in 2015-2017. Photographs were taken at the end of the late summer sampling periods each year. 2015 photos were taken in September of that year. 2016 and 2017 photos were taken in August of each year. Posts are missing from some 2017 photographs because they were broken by cattle. Q2 plots were not marked by posts.

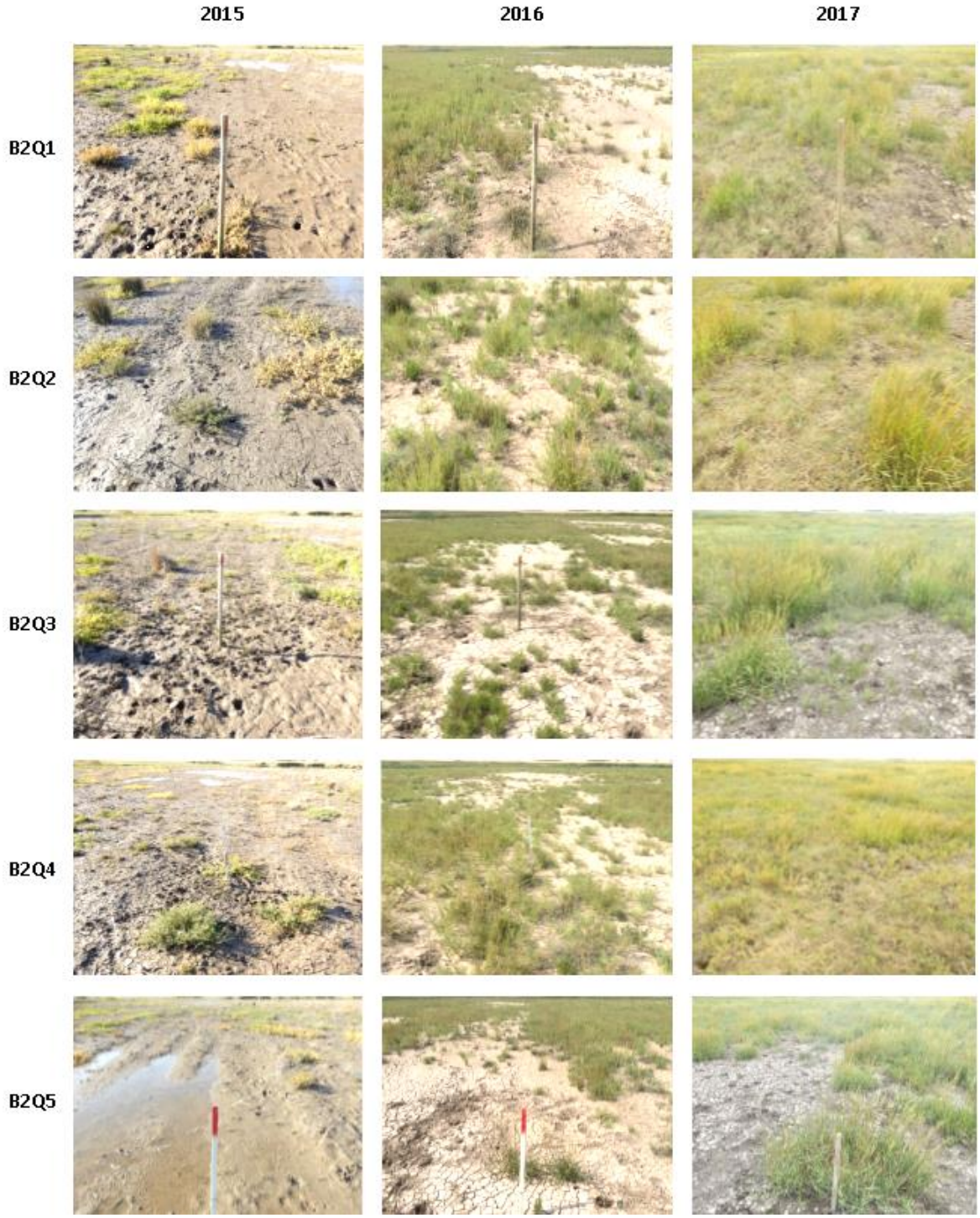


Figure 4.24 Photographs of plant succession on B2 quadrats in 2015-2017. Photographs were taken at the end of the late summer sampling periods each year. 2015 photos were taken in September of that year. 2016 and 2017 photos were taken in August of each year. Posts are missing from some 2017 photographs because they were broken by cattle. Q2 plots were not marked by posts.

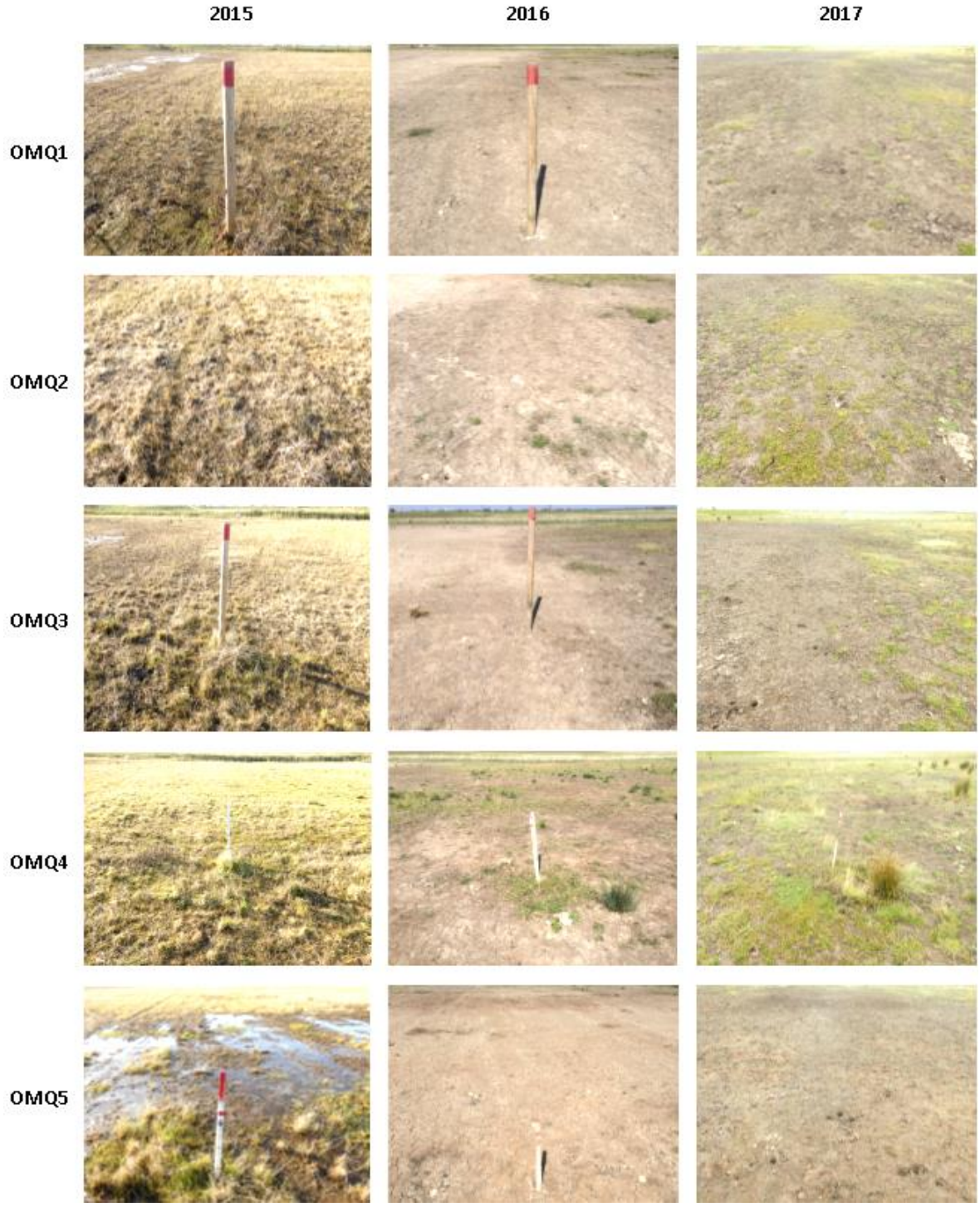


Figure 4.25 Photographs of plant succession on OM quadrats in 2015-2017. Photographs were taken at the end of the late summer sampling periods each year. 2015 photos were taken in September of that year. 2016 and 2017 photos were taken in August of each year. Posts are missing from some 2017 photographs because they were broken or displaced by cattle. Q2 plots were not marked by posts.

4.3.2.5 August 2018 vegetation succession and species richness

In August 2018 the study sites were revisited to check for levels of compaction in the quadrats (see Chapter 3). Final observations of vegatation coverage were taken during this period.

Bare ground was substantially less than it previously had been on A1, and also less on some A2 quadrats (A2Q3 and A2Q4). The species with the highest coverage was *P. maritima*, but there were also large stands of *A. tripolium* and *S. anglica*. *S. europeae* was found on these plots, but coverage was minimal. There was evidence that A. *prostrata* and *S. maritima* was still growing on these plots, but coverage was substatially lower than it had been in the initial stages post-breach (Figure 4.26).

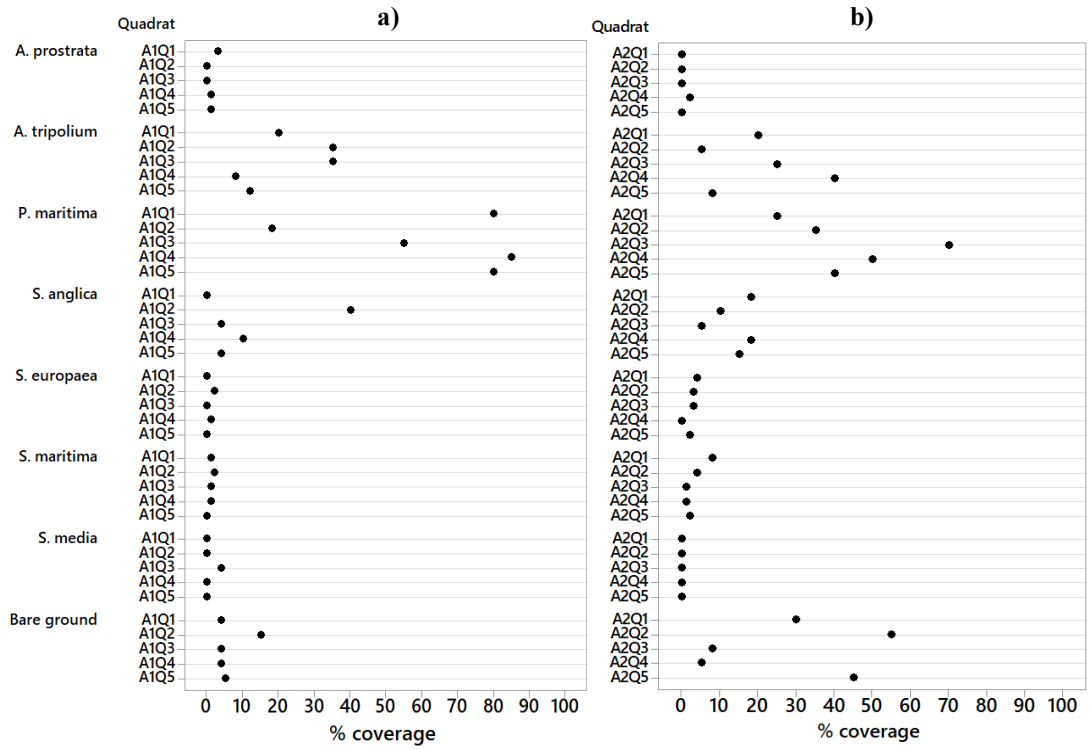


Figure 4.26 Vegetation coverage in each quadrat in a) A1 and b) A2 in August 2018.

By August 2018, all B1 and B2 quadrats contained at least some vegetation, but this was very minimal in B1Q4 and B1Q5, which still consisted predominately of bare ground. These quadrats had been under water for most of the study. *P. maritima* had grown further, and there were large stands of *S. anglica*. There was marginally more coverage of *A. tripolium* than the previous year, which was found in B1Q2 and B2Q2. Coverage of *S. europaea* was relatively high in B1Q2 and B1Q3 as it had been the previous year. *A. prostrata* coverage was very minimal, and *S. maritima* coverage was lower than the previous year (Figure 4.27).

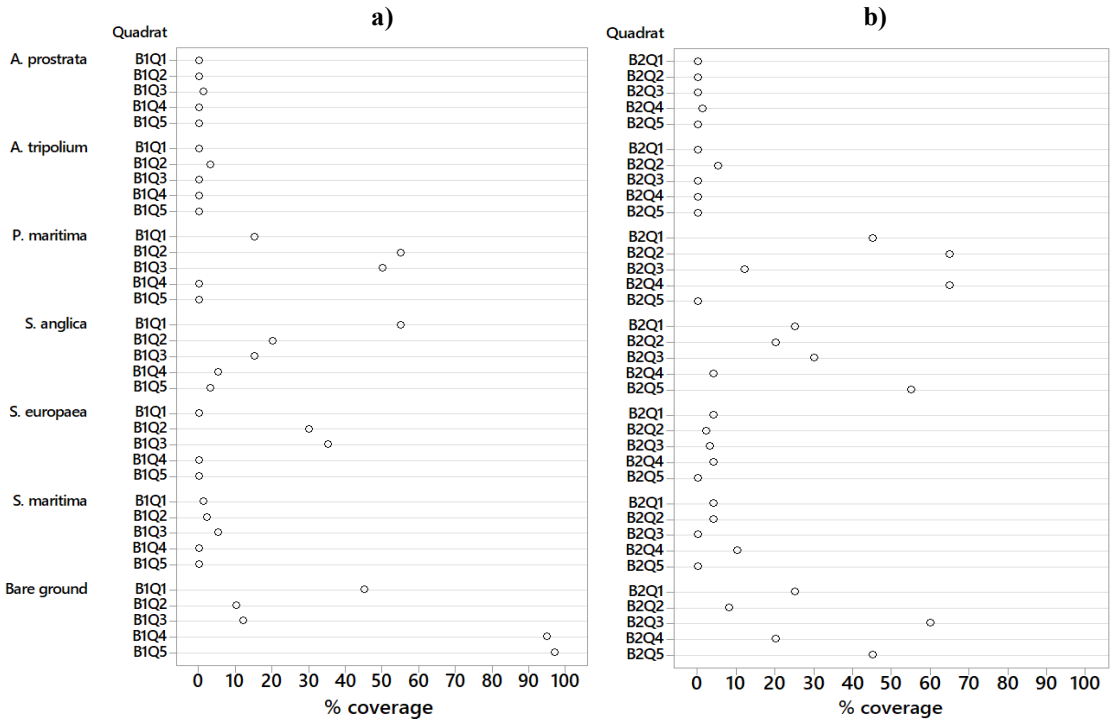


Figure 4.27 Vegetation coverage in each quadrat in a) B1 and b) B2 in August 2018.

On OM, *J. effusus* was the only species remaining from initial sampling pre-breach in August 2014. This was still on the edge of OMQ4, although it had been grazed in the previous years. There was some coverage of *A. prostrata* and *S. maritima* on this plot, and there was less bare ground than there had been the previous year. This is mainly due to the proliferation of *S. marina* and, to a lesser extent, *S. media* (Figure 4.28)

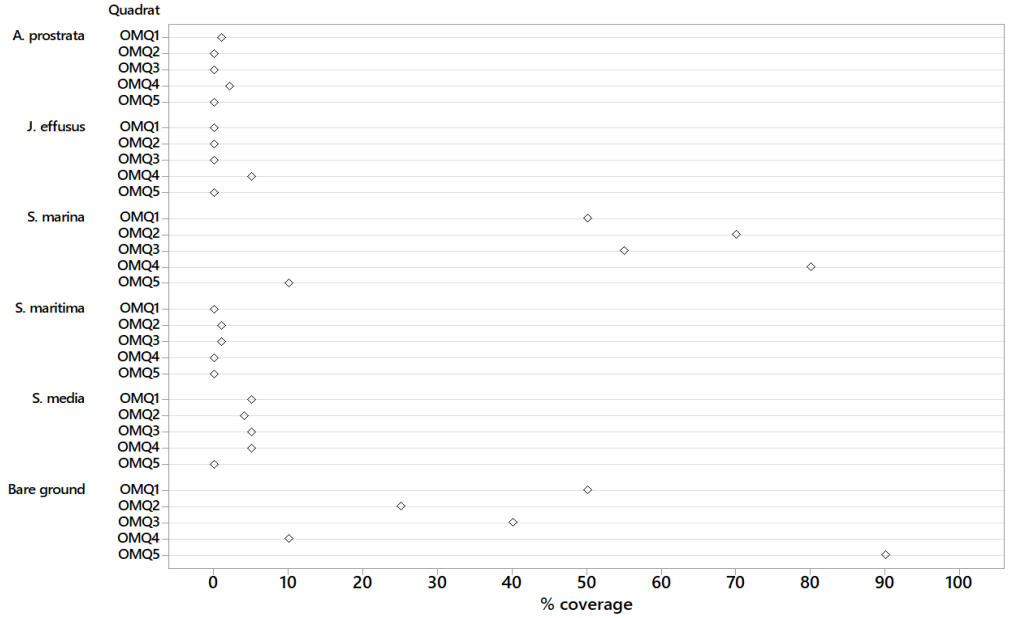


Figure 4.28 Vegetation coverage in OM quadrats in August 2018.

Over the duration of the study (2014-2018), species richness changed from being more species rich than the SSSI pre-breach (August 2014) to less species rich in the years

following breaching. B1 had the most species pre-breach, followed by B2, A1, A2 and OM. The decrease in species richness on OM was slower because existing vegetation remained longer on this site. It is important to note that green algae are classed as one species in descriptive data, but this could have been a number of different species. Of the plots on Steart Marsh, the highest number of species recorded was on A1 in 2018. The lowest number of different species recorded by 2018 was on OM (Figure 4.29).

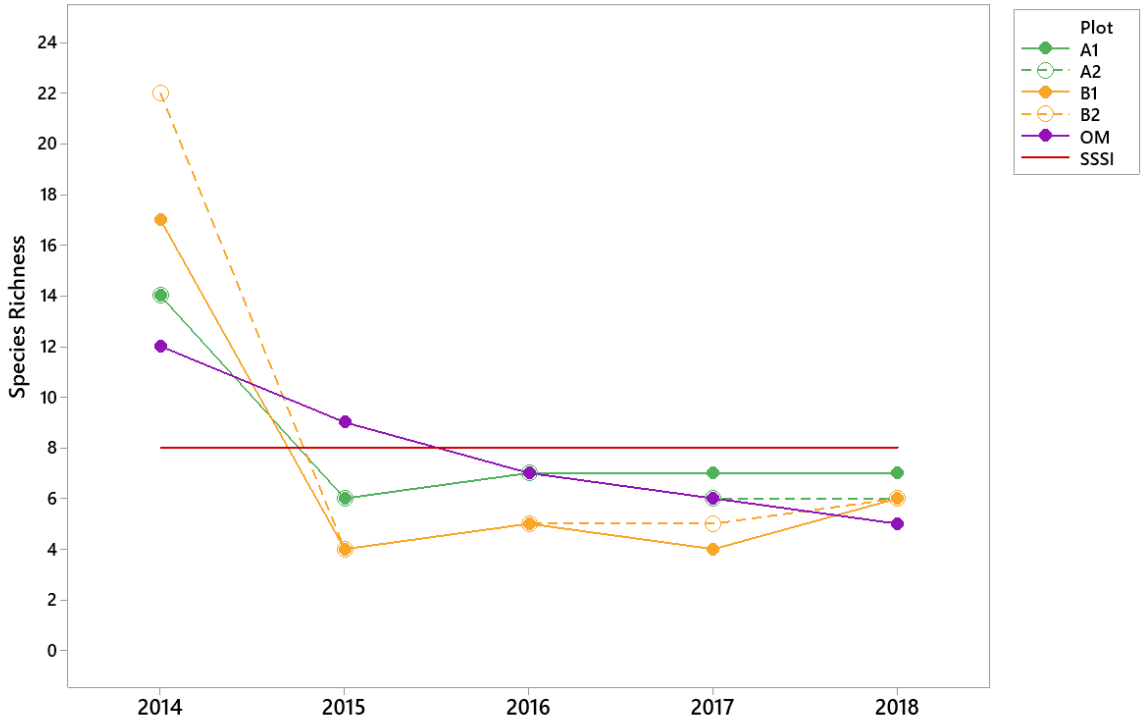


Figure 4.29 Species richness at the height of the growing season from August 2014 (pre-breach) to August 2018. For the purposes of these data, green algae spp. are counted as one species.

Data showed that the first two species to colonise Steart Marsh in April 2015 were an annual species (*A. prostrata*) and a short-lived perennial species (*C. anglica*). This was followed in August 2015 by a further annual species (*S. maritima*) and another short-lived perennial (*A. tripolium*), and perennial grasses had also started to colonise (Table 4.3). C. *anglica* was not found on the study plots in 2016, but the annual species, *S. europaea* was first found on these plots in 2016. Annual species continued to grow on Steart Marsh, but the perennial grasses had reduced the space that was available and coverage of annual species such as *A. prostrata* had declined by 2018.

Table 4.3 Colonisation of plants on Steart Marsh and Otterhampton Marsh from 2015-2018 related to life history traits. * Can be annual to short-term perennial depending on environmental factors.

Year	Scientific name	Life history	Plots
April 2015	*Atriplex prostrata*	Annual	A1, A2
	Cochlearia anglica	Short-lived perennial*	A1, A2
August 2015	*Atriplex prostrata*	Annual	A1, A2, B1, B2
	Suaeda maritima	Annual	A1, A2, B1, B2
	Cochlearia anglica	Short-lived perennial*	A1, A2
	Aster tripolium	Short-lived perennial*	A1, A2
	Puccinellia maritima	Perennial	A1, A2, B1, B2
	Spartina anglica	Perennial	A1, A2, B1, B2
2016	*Atriplex prostrata*	Annual	A1, A2, B1, B2
	Suaeda maritima	Annual	A1, A2, B1, B2
	Aster tripolium	Short-lived perennial*	A1, A2
	Puccinellia maritima	Perennial	A1, A2, B1, B2
	Spartina anglica	Perennial	A1, A2, B1, B2
	Salicornia europaea	Annual	A1, A2, B1, B2
	Spergularia marina	Annual	OM
2017	*Atriplex prostrata*	Annual	A1, A2
	Suaeda maritima	Annual	A1, A2, B1, B2, OM
	Aster tripolium	Short-lived perennial*	A1, A2, B2
	Puccinellia maritima	Perennial	A1, A2, B1, B2
	Spartina anglica	Perennial	A1, A2, B1, B2
	Salicornia europaea	Annual	A1, A2, B1, B2
	Spergularia marina	Annual	OM
	Spergularia media	Perennial	A1, OM
2018	*Atriplex prostrata*	Annual	A1, A2, B1, B2, OM
	Suaeda maritima	Annual	A1, A2, B1, B2, OM
	Aster tripolium	Short-lived perennial*	A1, A2, B1, B2
	Puccinellia maritima	Perennial	A1, A2, B1, B2
	Spartina anglica	Perennial	A1, A2, B1, B2
	Salicornia europaea	Annual	A1, A2, B1, B2
	Spergularia marina	Annual	OM
	Spergularia media	Perennial	A1, OM

4.3.3 NVC data

Vegetation coverage data were gathered on the SSSI in August 2016. The top three matches on the SSSI were SM12 (57.1%), SM6 (54.4%) and SM10 (49.6%) on Match and S21b (49.1%), SM10 (46.9%) and SM12 (44.4%) on MAVIS (Table 4.4).

Table 4.4 Top three flora community matches with NVC communities on the SSSI in August 2016. Community matches are derived from both MATCH and MAVIS software to provide a comparison between the two programs. The community match coefficient is in parentheses after each community code. Light blue shading represents saltmarsh communities; dark blue shading represents swamp communities.

	Closest NVC matches in August 2016	
Plot	**Match**	**MAVIS**
SSSI	1. SM12 (57.1%)	1. S21b (49.1%)
	2. SM6 (54.4%)	2. SM10 (46.9%)
	3. SM10 (49.6%)	3. SM12 (44.4%)

SM12 is a community dominated by *Aster tripolium*, SM6 is a *Spartina anglica* community, and SM10 is a transitional low-marsh vegetation community that is dominated by *Puccinellia maritima*, *Salicornia* and *Suaeda maritima.* S21b is an *Atriplex prostrata* subcommunity, so it is not surprising that this has been calculated on MAVIS due to the relatively high coverage of this species in some of the quadrats. However, SM12 was reported by both programs, and has a coefficient value > 50% on Match. Rodwell (2000) suggests that SM12 communities are found in this region of the Severn Estuary (Figure 4.30), so it is reasonable to suggest that this is a reliable match for the SSSI.

Map has been removed from this version of the thesis due to copyright restrictions

Figure 4.30 Location of SM11 and SM12 communities recorded in Great Britain. Map inset shows that SM12 rayed *Aster tripolium* stands have been recorded in Bridgwater Bay (Modified from Rodwell, 2000). Photograph inset shows *Aster tripolium* found on Steart Marsh in August 2017 (photograph taken by A. George, 2017).

The sample areas on Steart Marsh and Otterhampton Marsh prior to the breach (August 2014) most closely matched Mesotrophic Grassland (MG) communities, as expected, although Open Vegetation (OV) communities were reported on MAVIS for the Site B plots and OM (Table 4.5).

Table 4.5 Top three flora community matches with NVC communities on each sample plot at Steart Marsh and Otterhampton Marsh in August 2014. Community matches are derived from both Match and MAVIS software to provide a comparison between the two programs. The community match coefficient is in parentheses after each community code. Light green shading represents mesotrophic grassland communities and dark green shading represents vegetation of open habitats.

	Closest NVC matches August 2014	
Plot	**Match**	**MAVIS**
A1	1. MG6 (46.6%) 2. MG7b (42.7%) 3. MG7d (42.3%)	1. MG6c (49.7%) 2. MG6a (49.7%) 3. MG11a (48.7%)
A2	1. MG6 (38.3%) 2. MG9 (36.5%) 3. MG7b (36.2%)	1. MG11a (43.4%) 2. MG6a (43.1%) 3. MG10a (38.8%)
B1	1. MG6 (35.9%) 2. MG7b (35.3%) 3. MG11 (35.0%)	1. OV22b (43.5%) 2. OV19d (41.8%) 3. OV23c (41.6%)
B2	1. MG7b (38.4%) 2. MG6 (38%) 3. MG7a (37.4%)	1. OV22b (44.33%) 2. OV23c (42.98%) 3. OV23 (39.68%)
OM	1. MG11 (36.8%) 2. MG10 (35.2%) 3. MG13 (33.1%)	1. OV19d (44.7%) 2. OV20 (40.88%) 3. MG10a (40.66%)

The communities on Site A plots were matched more closely to MG communities at the outset because they were previously permanent pasture where there was a greater proportion of grass species. Site B was more disturbed and previously consisted of an arable crop rotation. As noted, there were more ruderal species on Site B that had colonised due to the disturbance during the initial phases of site creation, and the area was more compacted from the heavy machinery that was used to conduct the earthworks. This may be a reason why MAVIS matched these sites with OV communities, especially because these communities are predominantly dominated by weeds on disturbed grounds. However, at this stage all the coefficients were below 50% on both programs, indicating that the community matches were poor. Although homogenous plots were selected, the low coefficients were not unexpected due to the amount of disturbance these sites had prior to data collection.

Post breach, there was a change in the flora communities. The closest match on Site A plots in April 2015 was S21, which is a *Scirpus maritimus* swamp community (Table 4.6). Although this species was not found in the sample plots, it is clear why a swamp community was matched, because they are habitats with species poor vegetation. Indeed, much of the sample plots consisted of a high proportion of bare ground, with few species present. All four sample areas matched SM28. This community is *Elymus repens* salt-marsh community.

This was matched because of the presence of *A. prostrata* and *A. stolonifera*, which appear at a high frequency in these communities. Remnants of *A. stolonifera* were still present on the site from the baseline population, which was to be expected as populations can survive quite readily in waterlogged soil (Ahmad and Wainwright, 1977); however, this grass species was covered in sediment brought in by the tide. *A. prostrata* had also colonised the plots and was found on the edges of the banks. Moreover, *C. anglica* also began to colonise the plots, so it was clear that pioneer salt-marsh species were being brought in, potentially from the nearby SSSI.

Table 4.6 Top three flora community matches with NVC communities on each sample plot at Steart Marsh and Otterhampton Marsh in April 2015. Community matches are derived from both Match and MAVIS software to provide a comparison between the two programs. The community match coefficient is in parentheses after each community code. Light blue shading represents saltmarsh communities, dark blue shading represents swamp communities, light green shading represents mesotrophic grassland communities, dark green shading represents open vegetation communities, and yellow shading represents sand dune communities.

	Closest NVC matches April 2015	
Plot	**Match**	**MAVIS**
A1	1. S21 (40.3%)	1. S18b (46.4%)
	2. S18 (36.0%)	2. S21 (34.5%)
	3. SM28 (32.5%)	3. S21c (34.0%)
A2	1. S21 (35.5%)	1. S21 (32.47%)
	2. SM28 (33.8%)	2. S18b (30.5%)
	3. S18 (31.4%)	3. S20 (29.4%)
B1	1. MG11 (32.0%)	1. MG11 (29.6%)
	2. SM28 (29.0%)	2. S18b (27.8%)
	3. S18 (23.8%)	3. MG11b (27.3%)
B2	1. S18 (26.1%)	1. S18b (24.2%)
	2. SD1 (25.9%)	2. SD1 (23.4%)
	3. S21 (25.2%)	3. S18 (22.5%)
OM	1. MG11 (41.2%)	1. OV19d (41.9%)
	2. MG13 (34.8%)	2. OV21 (41.4%)
	3. MG7b (34.7%)	3. MG11a (40.7%)

One-year post-breach there were similar matches to the SSSI on the sites, and data calculated on Match had coefficients > 50% for SM10 in A1; SM12, SM6 and SM10 in A2; SM6, SM8 and SM9 in B1; and SM6, SM11 and SM12 in B2. MG communities were matched on OM, but the coefficients were very low (Table 4.7). Although there were similarities between Match and MAVIS data, it was noted that the coefficients derived in MAVIS were all <50%. This is probably due to the inclusion of bare ground in the analysis. Data derived from MAVIS shows that S21b is the highest match on A1 and A2. *A. prostrata*

was an early coloniser of the site, and its abundance was relatively high in August 2015 (see section 4.3.2.1), which is a likely reason for this NVC match with an *A. prostrata* subcommunity.

Table 4.7 Top three flora community matches with NVC communities on each sample plot at Steart Marsh and Otterhampton Marsh in August 2015. Community matches are derived from both Match and MAVIS software to provide a comparison between the two programs. The community match coefficient is in parentheses after each community code. Light blue shading represents saltmarsh communities, dark blue shading represents swamp communities and light green shading represents mesotrophic grassland communities.

	Closest NVC matches August 2015	
Plot	**Match**	**MAVIS**
A1	1. SM10 (50.5%) 2. SM12 (42.2%) 3. SM6 (40.4%)	1. S21b (48.7%) 2. SM10 (45.3%) 3. SM13a (41.4%)
A2	1. SM12 (54.4%) 2. SM6 (51.6%) 3. SM10 (50.5%)	1. S21b (47.4%) 2. SM10 (39.5%) 3. SM12 (37.0%)
B1	1. SM6 (59.7%) 2. SM8 (57.6%) 3. SM9 (42.7%)	1. SM6 (37.7%) 2. SM14a (36.6%) 3. SM8 (34.1%)
B2	1. SM6 (64.9%) 2. SM11 (58.7%) 3. SM12 (57.6%)	1. SM14a (49.7%) 2. SM14 (44.9%) 3. SM10 (43.27%)
OM	1. MG7b (32.6%) 2. MG6 (31.6%) 3. MG7e (27.8%)	1. MG7b (30.5%) 2. MG6 (30.0%) 3. MG6a (28.9%)

Community matches had not changed substantially by April 2016, although SM12 was the top match in A1 in Match and the second highest in MAVIS. S21b was still reported in MAVIS as the top matches on the Site A plots, but it was also reported on the Site B plots. This is likely partly due to the presence of small stands of *A. prostrata*. At this stage, OM was still in the stage of MG or OV communities, but the matches were very low (Table 4.8). There was still a lot of bare ground on the sites at this stage, and the annual plants, *A. prostrata* and *S. maritima* were in the stage of re-growth, so their absence in some quadrats may have had an influence on community matches.

Table 4.8 Top three flora community matches with NVC communities on each sample plot at Steart Marsh and Otterhampton Marsh in April 2016. Community matches are derived from both Match and MAVIS software to provide a comparison between the two programs. The community match coefficient is in parentheses after each community code. Light blue shading represents saltmarsh communities, dark blue shading represents swamp communities, light green shading represents mesotrophic grassland communities, and dark green shading represents open vegetation communities. Blue-green algae spp. was not included in the analysis because it is not a specific category on the NVC programs.

	Closest NVC matches April 2016	
Plot	**Match**	**MAVIS**
A1	1. SM12 (49.0%) 2. SM10 (39.6%) 3. SM6 (39.4%)	1. S21b (44.2%) 2. SM12 (37.9%) 3. SM13a (33.9%)
A2	1. SM6 (46.2%) 2. SM12 (42.5%) 3. SM8 (38.5%)	1. S21b (45.8%) 2. S4d (32.7%) 3. SM12 (30.7%)
B1	1. SM6 (51.9%) 2. SM12 (44.3%) 3. SM11 (37.3%)	1. SM6 (26.9%) 2. S21b (26.3%) 3. S4d (25.6%)
B2	1. SM6 (60.6%) 2. SM8 (38.0%) 3. S21 (38.0%)	1. S21b (34.0%) 2. S4d (31.8%) 3. SM6 (28.8%)
OM	1. MG7a (33.2%) 2. MG7b (18.2%) 3. MG7d (17.1%)	1. MG7a (24.2%) 2. OV18b (17.5%) 3. OV21b (17.3%)

By August 2016 percentage matches with communities was much higher on both Match and MAVIS. SM10 was the top match on both A1 (76.2% Match; 67.1% MAVIS) and on A2 (66.0% Match; 57.0% MAVIS). SM12 was also in the top three matches on both programs on both Site A plots (Table 4.9). SM10 is a traditional low-marsh vegetation community with *P. maritima*, *S. maritima* and *Salicornia* spp. as constants. By this stage the perennial *P. maritima* stands had grown in the quadrats and *S. europaea* was present. *S. maritima* was also a constant in all quadrats. *A. tripolium* was still relatively abundant in these plots, which is why SM12 was also a possible match.

Although moderately low in coverage, there were several *S. europaea* plants in the B1 quadrats (see section 4.3.2.2), which is why the plot matches with SM8, an annual *Salicornia* saltmarsh community. B2 plots also had *Salicornia* plants, but there was a higher abundance of other vegetation, including *S. anglica* and *P. maritima* in these quadrats. Therefore SM6 (*Spartina* community) and SM10 were also reported (Table 4.9). SM9, an *S. maritima* community, was also matched on B2 because of its relative abundance on the site. The matches were higher than that of the SSSI (top match: SM 12, 57.1%), but this may be

because there was less diversity on the Steart Marsh sample plots so the communities were matching more closely with the dominant species that are in the NVC communities, such as *P. maritima* on the Site A plots and *Salicornia* on the Site B plots.

The OM quadrats were essentially devoid of vegetation by August 2016, with four of the five quadrats having > 90% bare ground. A few remnants of existing vegetation remained, especially in OMQ4, which also had been colonised by one *S. marina* plant. Due to the lack of vegetation coverage across all quadrats it was decided that an NVC analysis could not be performed on these data.

Table 4.9 Top three flora community matches with NVC communities on each sample plot at Steart Marsh and Otterhampton Marsh in August 2016. Community matches are derived from both Match and MAVIS software to provide a comparison between the two programs. The community match coefficient is in parentheses after each community code. Light blue shading represents saltmarsh communities.

	Closest NVC matches August 2016	
Plot	**Match**	**MAVIS**
A1	1. SM10 (76.2%) 2. SM9 (66.7%) 3. SM12 (65.6%)	1. SM10 (67.1%) 2. SM12 (54.6%) 3. SM9 (52.8%)
A2	1. SM10 (66.0%) 2. SM6 (60.9%) 3. SM12 (59.6%)	1. SM10 (57.0%) 2. SM13a (45.8%) 3. SM12 (45.2%)
B1	1. SM8 (79.2%) 2. SM6 (67.1%) 3. SM11 (65.8%)	1. SM8 (56.7%) 2. SM10 (50.5%) 3. SM11 (49.4%)
B2	1. SM6 (70.7%) 2. SM9 (68.2%) 3. SM11 (65.8%)	1. SM9 (51.0%) 2. SM10 (50.4%) 3. SM8 (45.8%)
OM	Vegetation coverage too low to match with NVC communities.	

By April 2017, the top two matches on the Site A plots were SM12 and SM10, and the top two matches on B1 were SM6 and SM8. The top two matches on B2 were SM6 and SM10 according to Match and SM10 and SM14a (*Atriplex portulacoides*, formally *Halimione portulacoides*, community) according to MAVIS (Table 4.10).

As noted in section 4.3.2.3 permission to monitor OM was denied in the spring and early summer of April 2017 due to breeding birds on the site.

Table 4.10 Top three flora community matches with NVC communities on each sample plot at Steart Marsh and Otterhampton Marsh in April 2017. Community matches are derived from both Match and MAVIS software to provide a comparison between the two programs. The community match coefficient is in parentheses after each community code. Light blue shading represents saltmarsh communities.

	Closest NVC matches April 2017	
Plot	**Match**	**MAVIS**
A1	1. SM12 (64.5%) 2. SM10 (62.6%) 3. SM6 (57.0%)	1. SM10 (55.2%) 2. SM12 (48.5%) 3. SM13a (45.1%)
A2	1. SM12 (65.3%) 2. SM10 (63.2%) 3. SM6 (61.0%)	1. SM10 (54.3%) 2. SM12 (49.9%) 3. SM13a (49.2%)
B1	1. SM6 (70.8%) 2. SM8 (68.1%) 3. SM10 (67.5%)	1. SM8 (47.8%) 2. SM6 (45.3%) 3. SM14a (42.6%)
B2	1. SM6 (74.7%) 2. SM10 (62.7%) 3. SM12 (62.2%)	1. SM10 (52.6%) 2. SM14a (48.5%) 3. SM14 (44.4%)
OM	Data could not be gathered due to breeding birds on OM	

Community composition did not change greatly from April 2017 to August 2017, apart from the growth of plants that had been monitored in the spring. But it was clear that the community matches on the Site A plots were more similar to those of the SSSI than the other plots, although SM6 and SM10 were matched on the Site B plots (Table 4.11). By this stage more patches of *S. anglica* and *P. maritima* were found on the Site B plots and they were growing into large stands (see section 4.3.2.2).

Matches with communities were very low on OM, but SM23 was the highest as determined by both Match and MAVIS (Table 4.11). SM23 has *S. marina* as a constant species. Although there was very little coverage of vegetation on OM at this time, the species was present along with *S. media*.

By 2018, the highest matches on both Site A plots was SM12 on Match and SM10 on MAVIS, although SM12 was also matched at > 50% on MAVIS. SM6 was the highest match on the Site B plots, most likely due to the presence of *S. anglica*, although SM10 was the highest match for B2 on MAVIS. SM23 was still the highest match for OM communities, but the matches had coefficients <50% (Table 4.12).

Table 4.11 Top three flora community matches with NVC communities on each sample plot at Steart Marsh and Otterhampton Marsh in August 2017. Community matches are derived from both Match and MAVIS software to provide a comparison between the two programs. The community match coefficient is in parentheses after each community code. Light blue shading represents saltmarsh communities.

	Closest NVC matches August 2017	
Plot	**Match**	**MAVIS**
A1	1. SM10 (76.3%) 2. SM12 (74.5%) 3. SM6 (64.4%)	1. SM10 (65.0%) 2. SM12 (54.5%) 3. SM13a (38.4%)
A2	1. SM12 (73.6%) 2. SM10 (72.7%) 3. SM6 (64.1%)	1. SM10 (63.4%) 2. SM12 (54.8%) 3. SM13a (48.3%)
B1	1. SM6 (84.1%) 2. SM8 (68.3%) 3. SM9 (60.4%)	1. SM8 (46.7%) 2. SM6 (44.6%) 3. SM10 (42.3%)
B2	1. SM6 (79.0%) 2. SM8 (63.9%) 3. SM9 (62.7%)	1. SM10 (52.6%) 2. SM14a (48.5%) 3. SM9 (47.6%)
OM	1. SM23 (34.8%) 2. S21 (24.6%) 3. SM12 (23.0%)	1. SM23 (25.2%) 2. S21 (20.7%) 3. SM12 (20.0%)

Table 4.12 Top three flora community matches with NVC communities on each sample plot at Steart Marsh and Otterhampton Marsh in August 2018. Community matches are derived from both Match and MAVIS software to provide a comparison between the two programs. The community match coefficient is in parentheses after each community code. Light blue shading represents saltmarsh communities.

	Closest NVC matches August 2018	
Plot	**Match**	**MAVIS**
A1	1. SM12 (74.9%) 2. SM10 (64.8%) 3. SM6 (64.2%)	1. SM10 (56.1%) 2. SM12 (55.7%) 3. SM13a (49.1%)
A2	1. SM12 (73.6%) 2. SM10 (72.7%) 3. SM6 (68.6%)	1. SM10 (63.4%) 2. SM12 (54.8%) 3. SM13a (48.3%)
B1	1. SM6 (83.1%) 2. SM8 (64.1%) 3. SM12 (59.4%)	1. SM6 (46.5%) 2. SM14a (45.6%) 3. SM8 (44.2%)
B2	1. SM6 (81.7%) 2. SM8 (70.2%) 3. SM12 (63.9%)	1. SM10 (51.5%) 2. SM8 (51.0%) 3. SM6 (48.0%)
OM	1. SM23 (38.8%) 2. S21 (28.9%) 3. SM12 (26.0%)	1. SM23 (35.4%) 2. S21 (25.3%) 3. SM12 (23.5%)

4.3.4 Cluster analysis and ordination

Cluster analysis was conducted to explore the similarities in quadrat species composition with the mature salt marsh (SSSI). Baseline data from August 2014 show groupings related to starting states (Figure 4.31). Apart from quadrat A2Q4, all Site A quadrats were grouped together, and most of the OM and B quadrats formed clusters with one another. The SSSI quadrats were clustered together separately. This was expected because the Site A plots were pasture at this stage and the Site B plots and OM were previously arable and had a higher percentage of bare ground and ruderal species. The SSSI quadrats were grouped together because they represented a saltmarsh community.

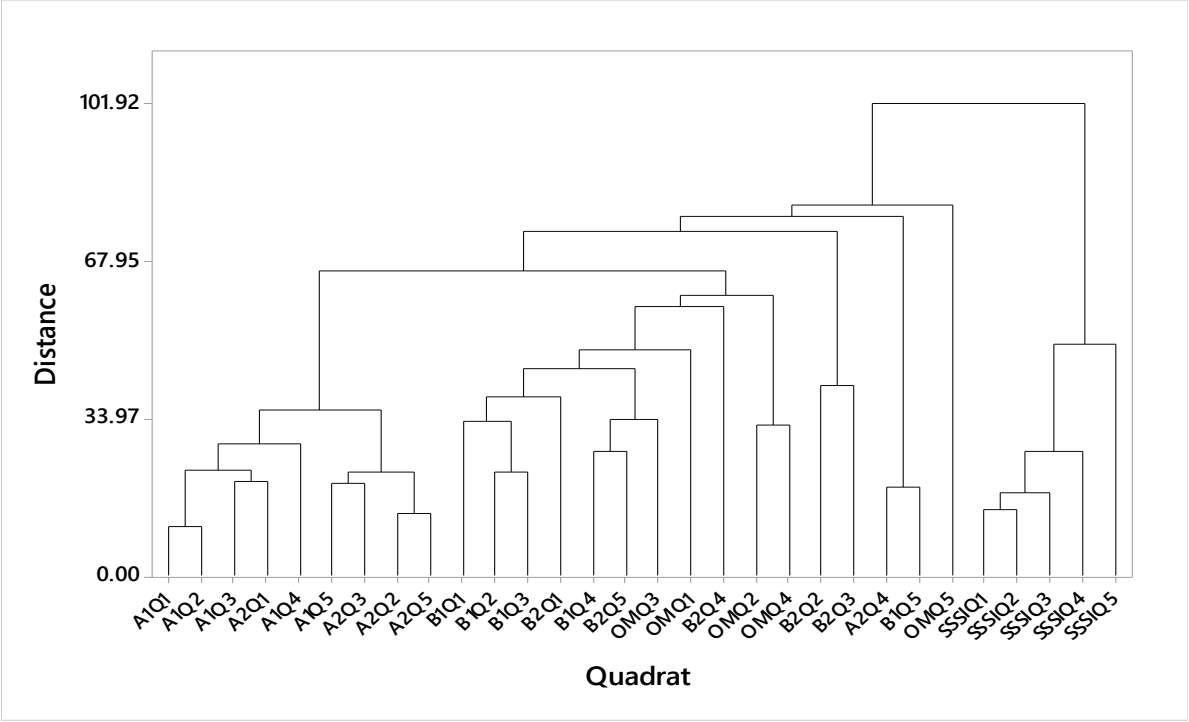


Figure 4.31 Clustering of quadrats in August 2014 (before the breach) using the average linkage method and Euclidean distance. SSSI quadrats were samples in August 2016.

Before the breach the starting states of the two main sites (A plots and B plots) were different and this can be clearly seen in the analysis (Figure 4.31). Nevertheless, they did have some common species that had colonised due to the disturbance and change in land-use during the construction period of the site. OM was generally like Site B, although there were some differences, such as the presence of rush (*Juncus effuses*) which was found in OMQ4. It is clear that these quadrats were very different in species composition from the SSSI at this time, because they had not been under the influence of a tidal regime.

Eigenvalues derived from DCA analysis of August 2014 vegetation coverage showed that both DCA1 and DCA2 axes were meaningful (eigenvalues ≥ 0.5). The species

scores show that in August 2014, the species were grouped according to original land-use. The terrestrial agricultural species were grouped and the chracteristic saltmarsh species were grouped. It is important to note, however, that the terrestrial species were not all grouped together on the DCA2 axis and had varying scores (Figure 4.32). At this stage the sites had many ruderal species, and these plots were not being managed as agricultural sites since the development of Steart Marshes began. This is why they were not forming tighter groups into charactersitic pasture species and arable species.

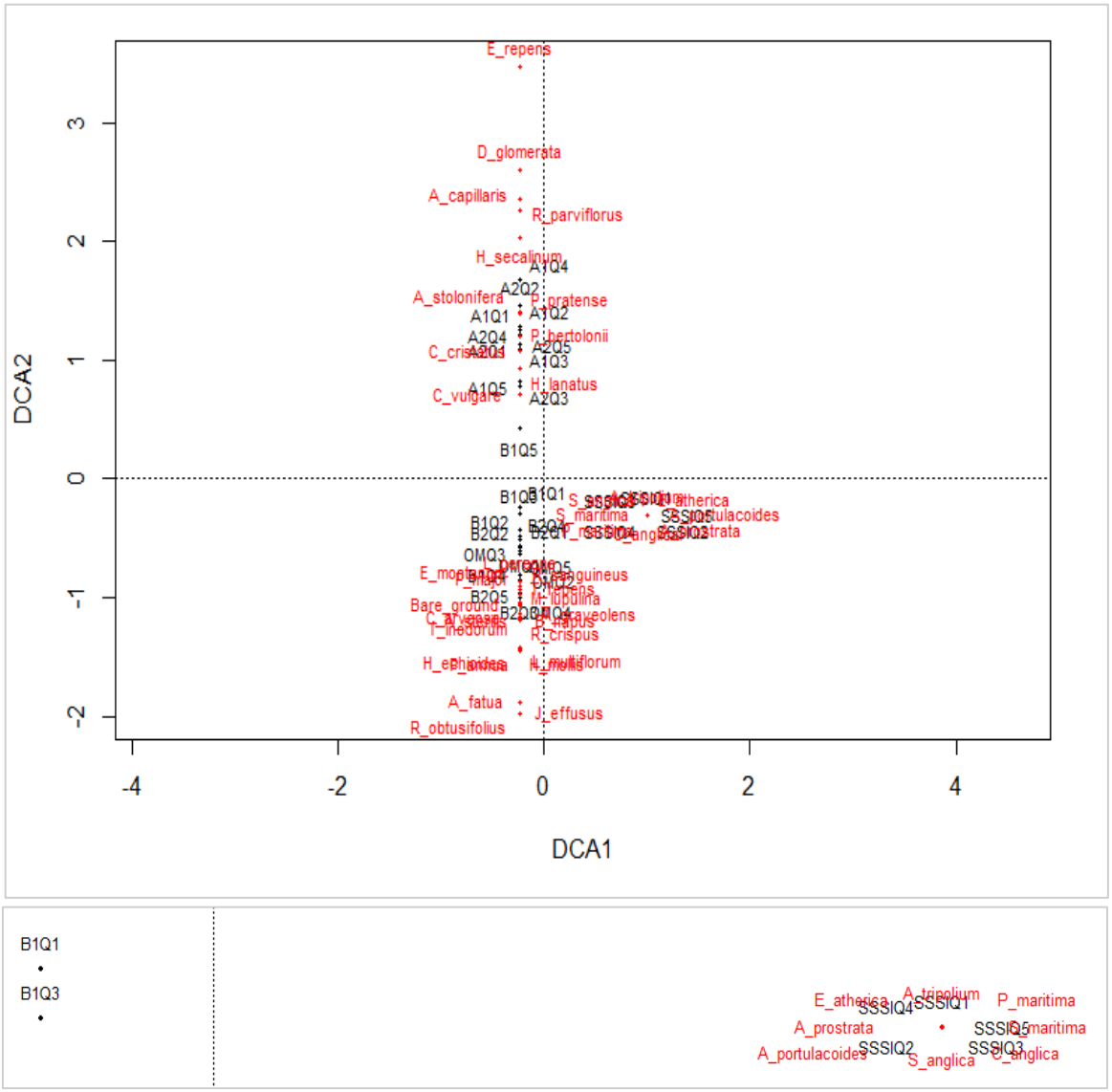


Figure 4.32 Detrended correspondence analysis biplot for August 2014 vegetation data showing groupings of quadrats and the species responsible. Quadrats are in black font and species are in red font. Inset highlights how different the SSSI is from other plots due to the presence of halophytic species. DCA1 had an eigenvalue of 1.0. DCA2 had an eigenvalue of 0.5. All other axes had eigenvalues < 0.5.

In April 2015, most of the quadrats on Steart Marsh formed tight clusters in analysis because there was a high level of bare ground in each of the quadrats (Figure 4.33). The OM

quadrats were marginally closer to the SSSI quadrats in this analysis, possibly because they had less bare ground than on Steart Marsh, which was like the SSSI. However, these OM quadrats were still a considerable distance from the SSSI in terms of vegetation coverage because OM had terrestrial agricultural and ruderal species, whereas the SSSI had characteristic saltmarsh vegetation.

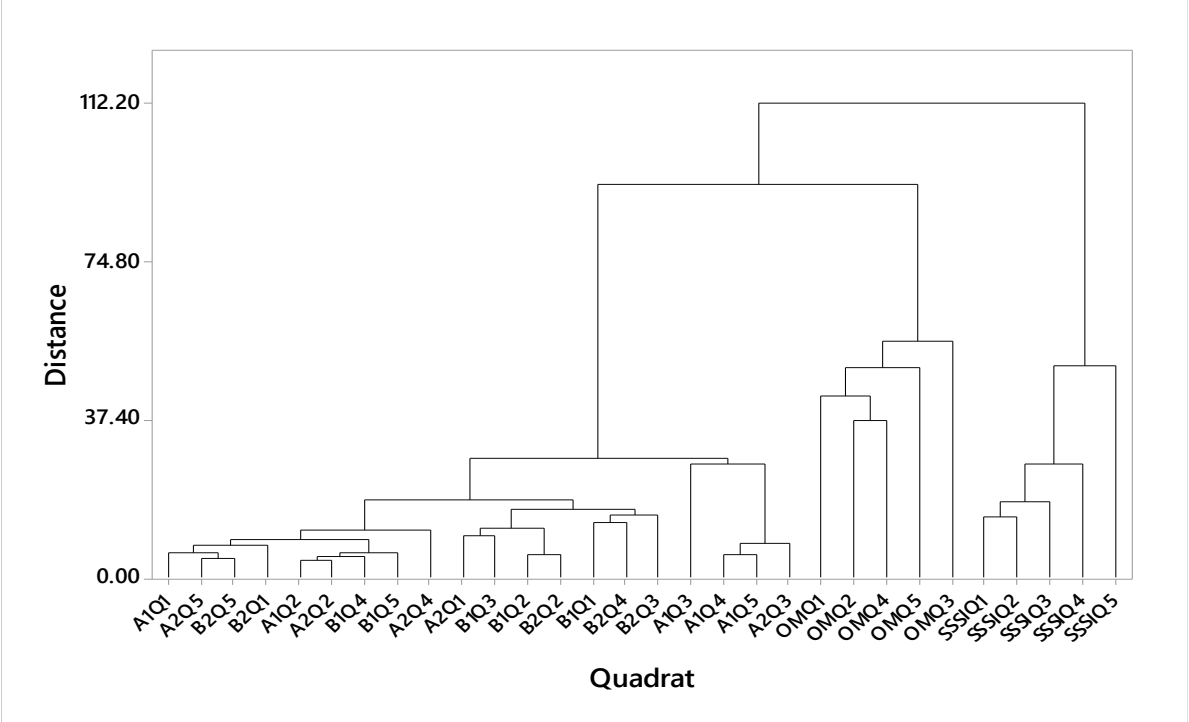


Figure 4.33 Clustering of quadrats in April 2015 (8 months after the breach) using the average linkage method and Euclidean distance.

The DCA1 axis was meaningful in April 15 (eigenvalue = 1.0), but all other axes had values <0.5. It was clear that there was still a distance between Steart Marsh quadrats and the SSSI, but as *C. anglica* and *A. prostrata* were found on the Site A plots, some Site A quadrats were closer to the SSSI, especially A1Q2 and A2Q5 (Figure 4.34). This analysis shows that OM quadrats are the furthest away from the SSSI because of the difference in vegetation across the environmental gradient on DCA1 (terrestrial to saltmarsh).

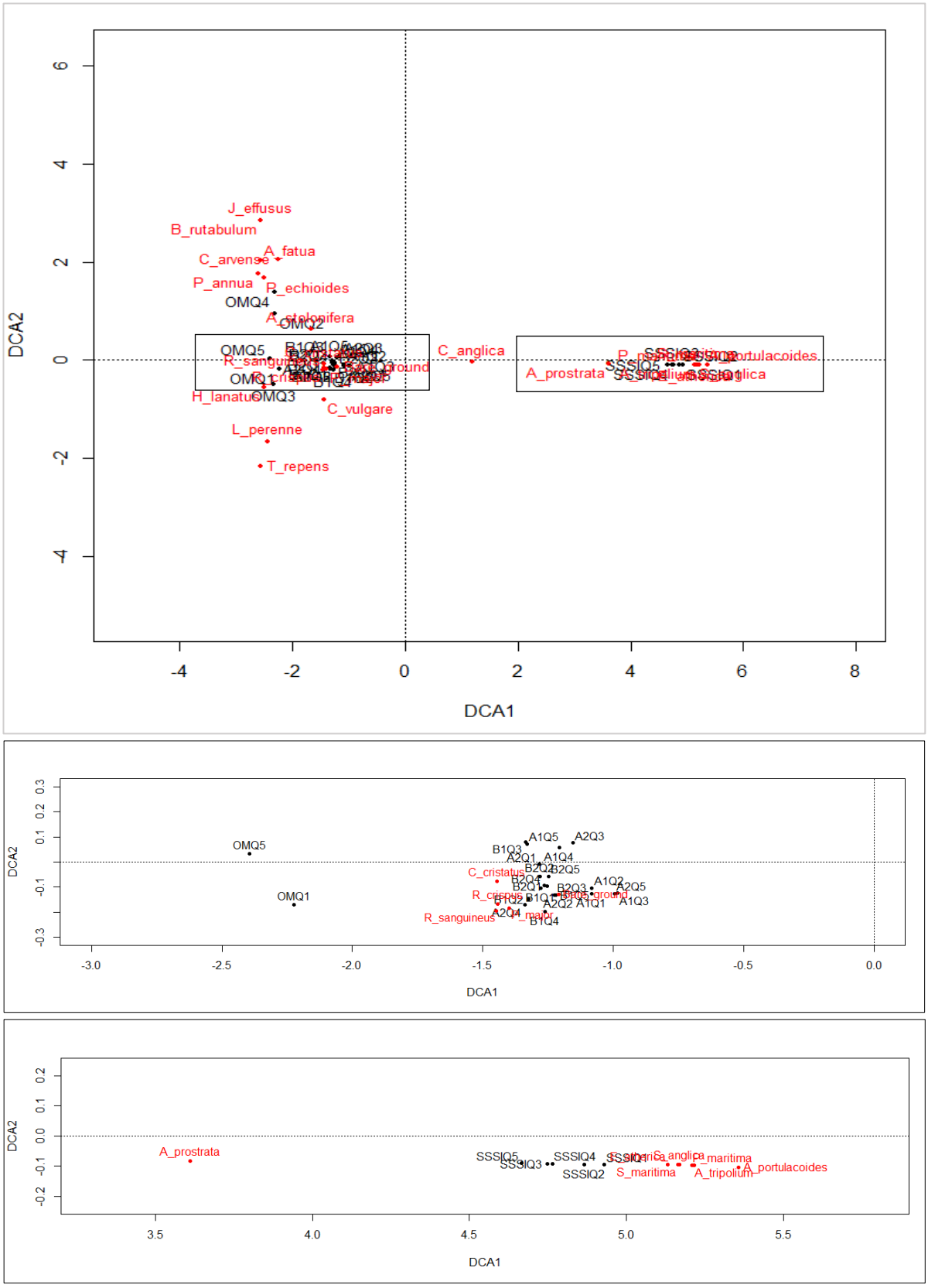


Figure 4.34 Detrended correspondence analysis biplot for April 2015 vegetation data showing groupings of quadrats and the species responsible. Quadrats are in black font and species are in red font. Inset images (i and ii) show magnification of the biplot. DCA1 had an eigenvalue of 1.0. All other axes had eigenvalues < 0.5.

In August 2015, there were some changes in groupings (Figure 4.35), but most Site A quadrats were still clustered together. There were tighter clusters between some of the quadrats at this stage, which was driven by the amount of bare ground. Therefore, some of the Site A quadrats were grouping more closely with the Site B quadrats. At this stage, the OM quadrats were generally clustered together, because they had not had the impact from the tidal regime that had occurred on Sites A and B.

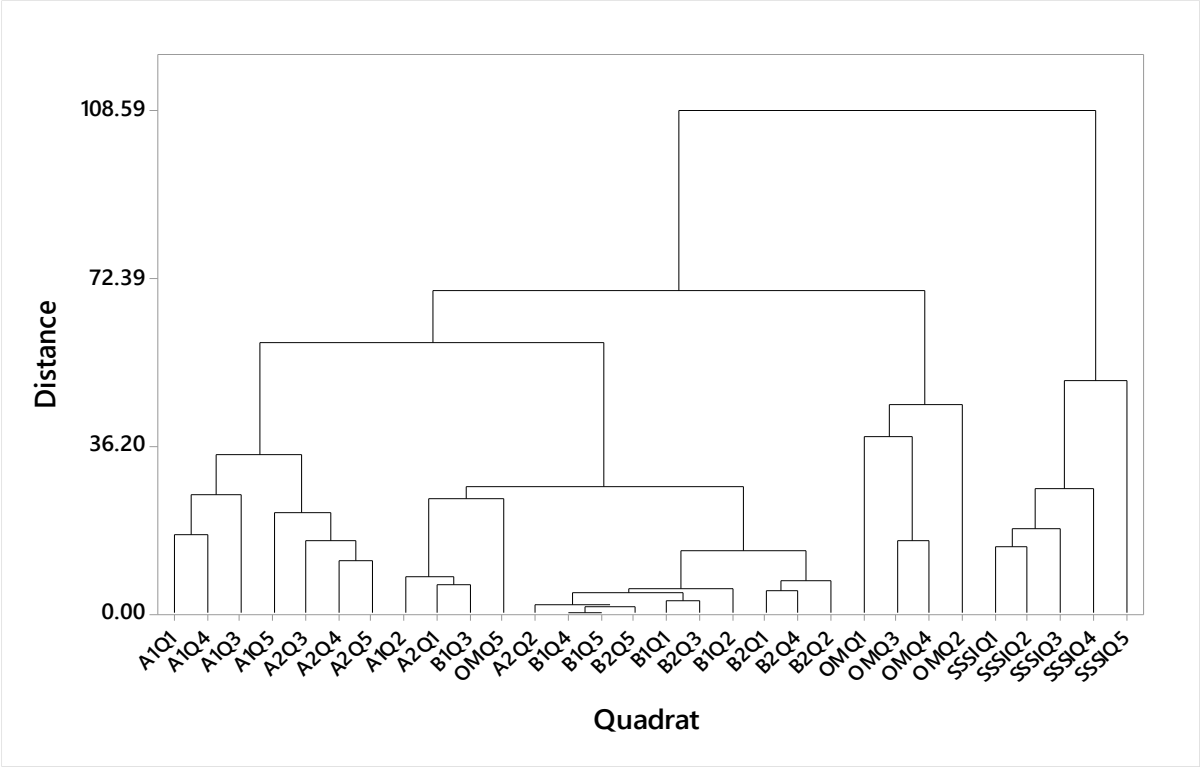


Figure 4.35 Clustering of quadrats in August 2015 (12 months after the breach) using the average linkage method and Euclidean distance.

Nevertheless, characteristic saltmarsh species had started to colonise the Site A quadrats more rapidly, which can be seen in the DCA (Figure 4.36). Most Site A quadrats were grouping together on DCA1, which was the only meaningful axis (eigenvalue = 0.8), and some quadrats had started to tend toward the SSSI sites. However, the site scores in the DCA analysis show that the quadrats remained relatively different to the SSSI. A1Q1 and A1Q3 had similar species that were also found on the SSSI, but other quadrats were grouped due to the prevalence of bare ground. A2Q2 was in a very similar state to the Site B quadrats because there was minimal vegetation coverage. The scores on DCA1 clearly showed that the species were still grouping by terrestrial and saltmarsh associations (Figure 4.35). However, both *S. maritima* and *A. prostrata* had closer scores to the terrestrial species. Bare ground had scores on the axis that were between both terrestrial and saltmarsh species.

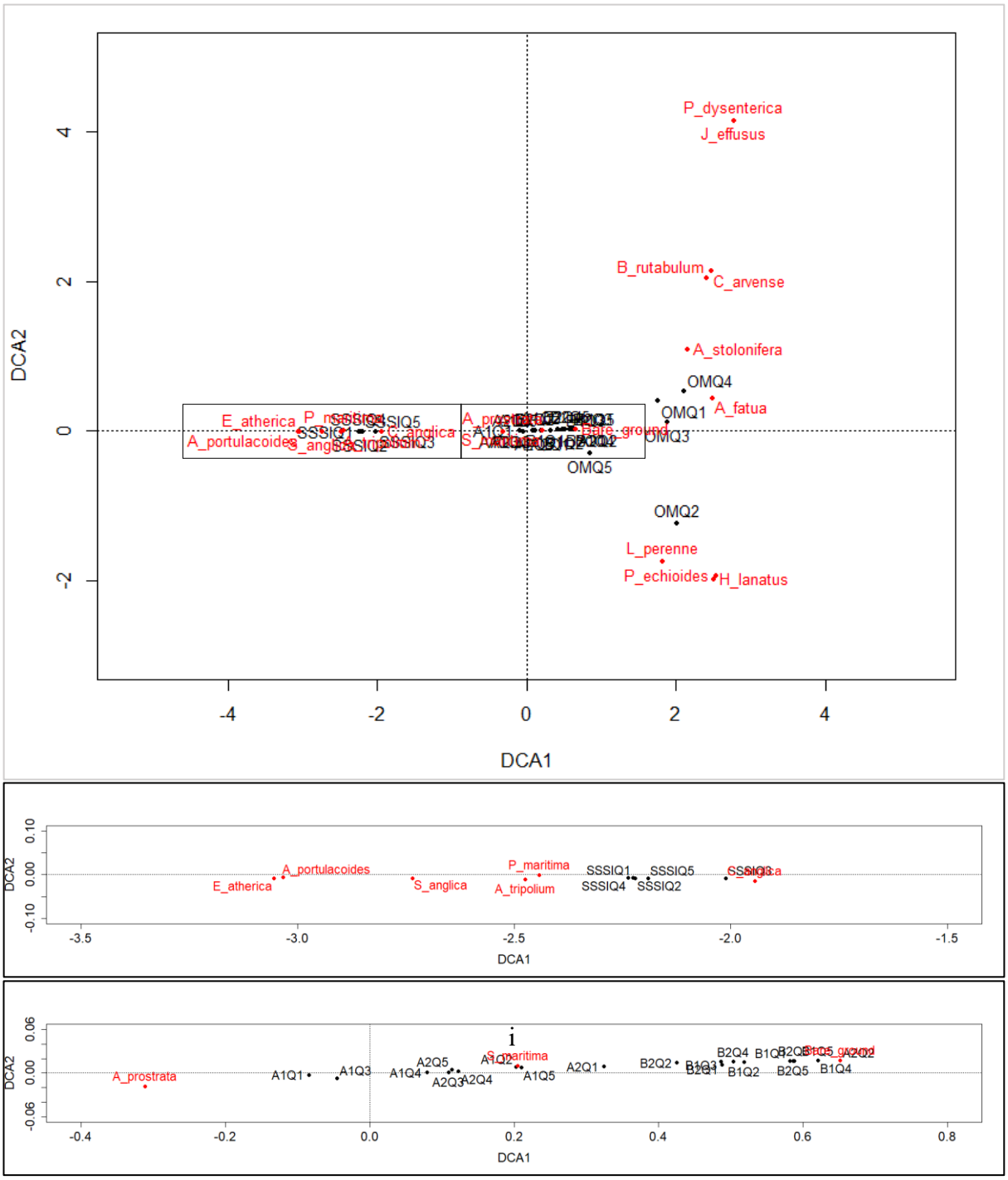


Figure 4.36 Detrended correspondence analysis biplot for August 2015 vegetation data showing groupings of quadrats and the species responsible. Quadrats are in black font and species are in red font. Inset images (i and ii) show magnification of the biplot. DCA1 had an eigenvalue of 0.8. All other axes had eigenvalues < 0.5.

In April 2016 one of the former pasture quadrats (A1Q3) had moved closer to the SSSI because of similarities in species (Figure 4.37). Bare ground was still having a major influence on these groupings, which can be seen in the DCA ordination (Figure 4.38) and, at this stage of the year, the vegetation was close to the ground on the sites because annual species, such as *A. prostrata* had died back, and new growth was beginning.

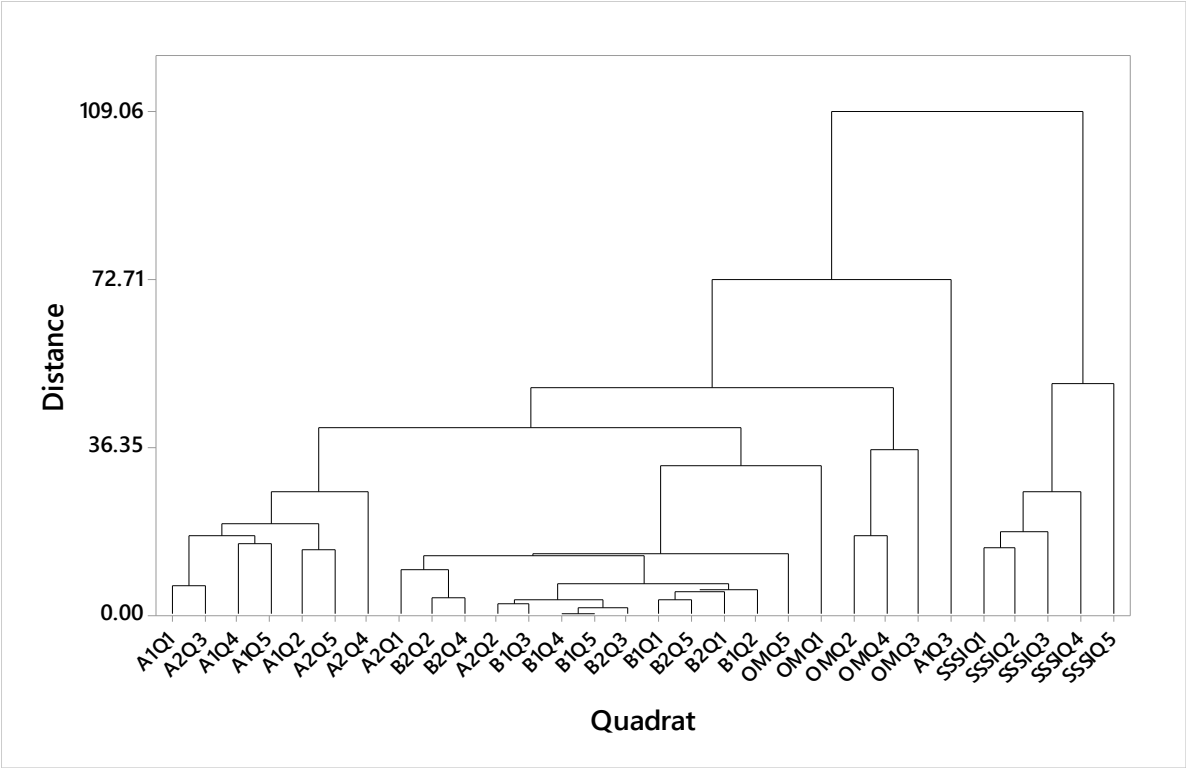


Figure 4.37 Clustering of quadrats in April 2016 (20 months after the breach) using the average linkage method and Euclidean distance.

The ordination shows that *A. tripolium* was starting to influence the groupings of some of the Site A quadrats by April 2016 (Figure 4.38). OM groupings were still being governed by species that were originally observed in Aug 2014, *L. perenne* and *J. effusus*, but also by the green algae that had been growing on the bare substrate. Bare ground was the main influence on the Site B quadrats.

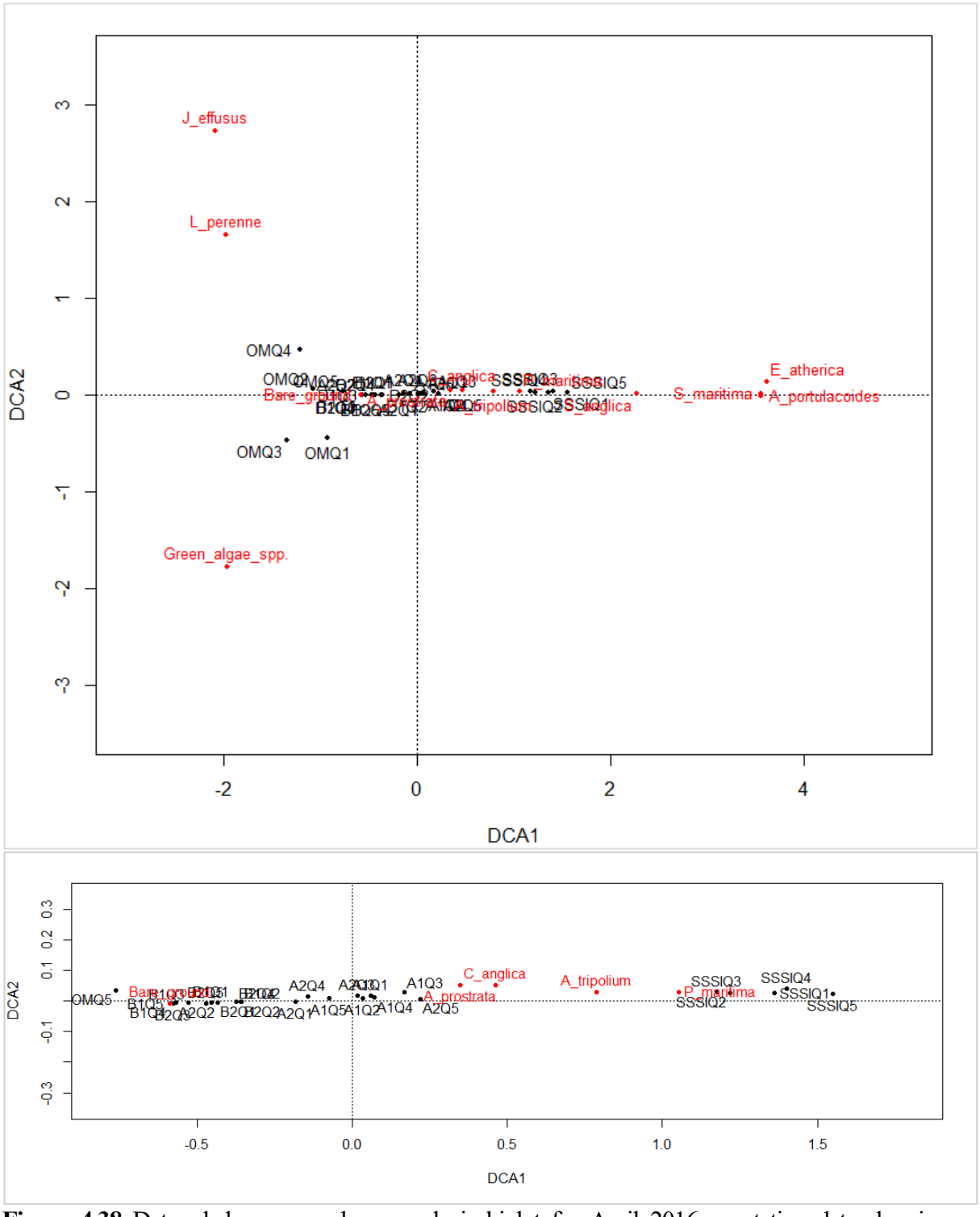


Figure 4.38 Detrended correspondence analysis biplot for April 2016 vegetation data showing groupings of quadrats and the species responsible. Quadrats are in black font and species are in red font. Smaller image shows magnification of the biplot. DCA1 had an eigenvalue of 0.6. All other axes had eigenvalues < 0.5.

In August 2016, the SSSI quadrats were closer in distance to most of the Site A quadrats because there were now similarities in species present (Figure 4.39). The differences existed because Site A plots had less diversity in the quadrats than the SSSI at this stage, and there was still bare ground in some of the quadrats. The OM and Site B

quadrats had still not established, which is why they were grouped together in some of the clusters.

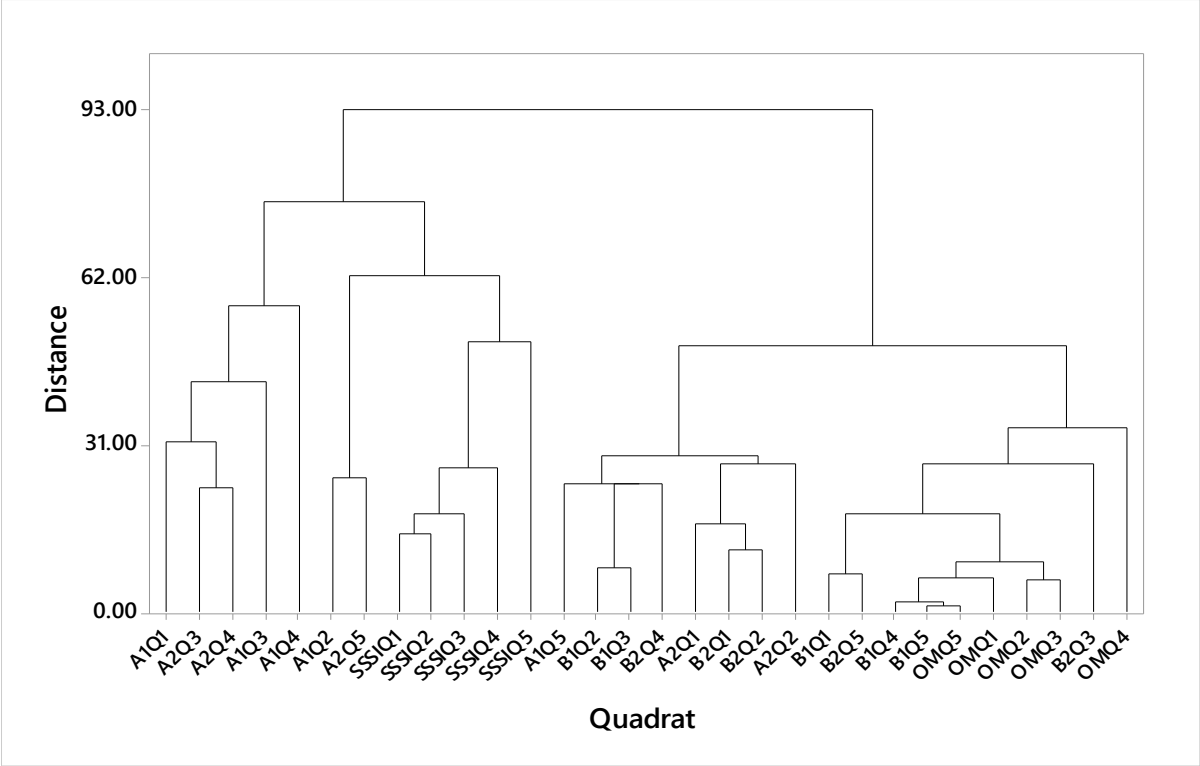


Figure 4.39 Clustering of quadrats in August 2016 (24 months after the breach) using the average linkage method and Euclidean distance.

The site scores and DCA scatterplots show how the study sites on Steart Marsh had changed considerably by August 2016 (Figure 4.40). There were some quadrats that were tending towards the SSSI. Some of the pasture quadrats had scores that were closer to the SSSI scores, which suggested that these plots were moving toward similar vegetation coverage and diversity quicker than the other plots, which corresponded with cluster analysis data. The OM quadrats were quite a distance from the SSSI because they had a large amount of bare ground and still had some characteristic terrestrial species. The arable quadrats (Site B) also had extensive bare ground, but they did have some saltmarsh plants colonising. Some of the pasture quadrats, such as A2Q2, were grouped with the arable sites because this quadrat still had no vegetation coverage. The terrestrial species were grouped closely together because these only remained on Otterhampton Marsh. Some of the saltmarsh species, such as *E. atherica* and *A. portulacoides* had very different scores and are separated on the scatterplot (Figure 4.40) because they were only found in some quadrats on the SSSI and not on any other site. *A. tripolium*, *A. prostrata* and *P. maritima* were important species at this stage because they had colonised Steart Marsh rapidly and were also found in

abundance on the SSSI. However, bare ground was still playing a key role at this stage of the site development.

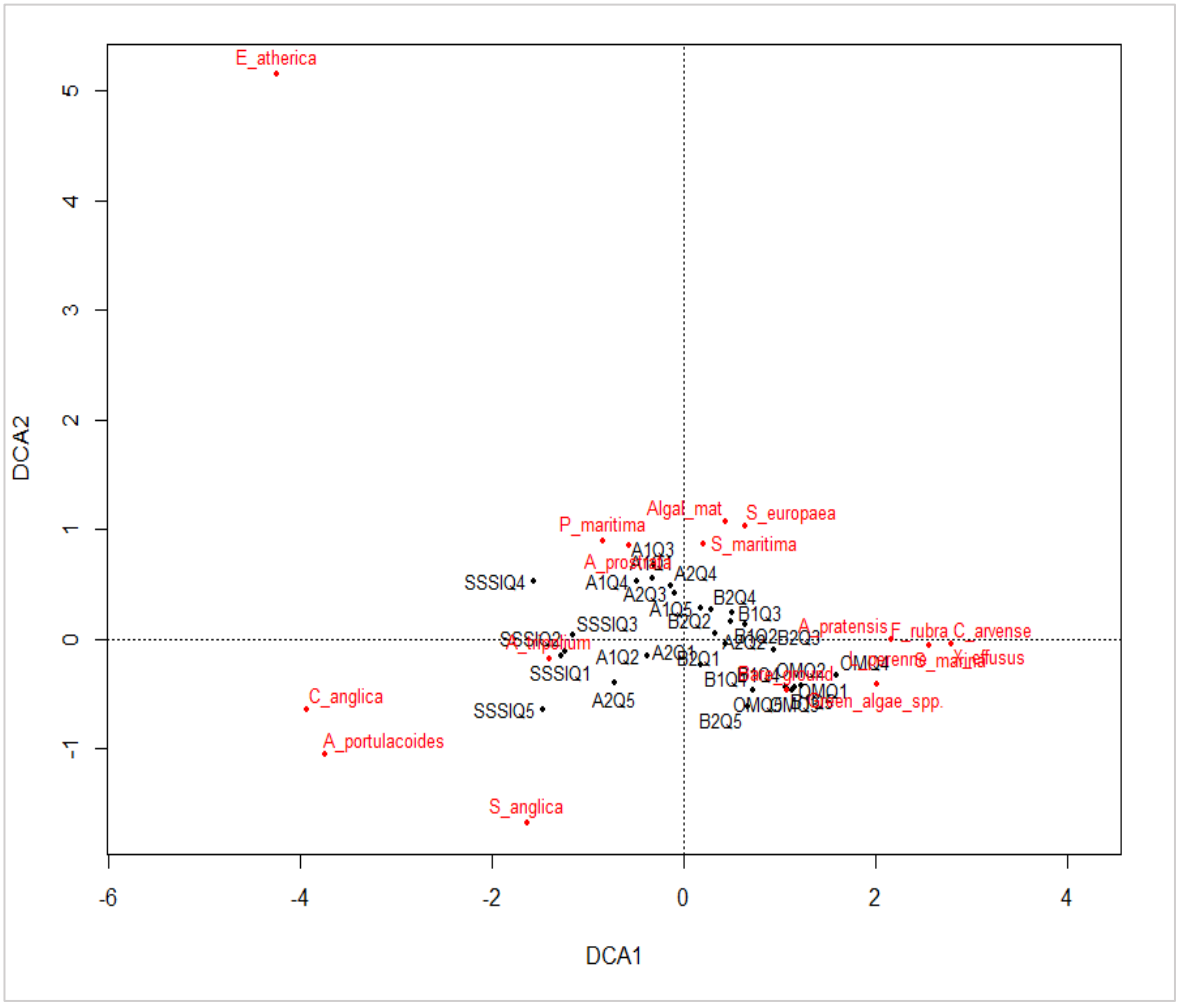


Figure 4.40 Detrended correspondence analysis biplot for August 2016 vegetation data showing groupings of quadrats and the species responsible. Quadrats are in black font and species are in red font. DCA1 had an eigenvalue of 0.6. All other axes had eigenvalues < 0.5.

April 2017 clustering showed that four of the A1 quadrats (A1Q1, A1Q3, A1Q4 and A1Q5) were closest in distance to the SSSI quadrats (Figure 4.41). As noted in section 4.3.2.3 OM quadrats could not be assessed because access to the site was denied due to breeding birds.

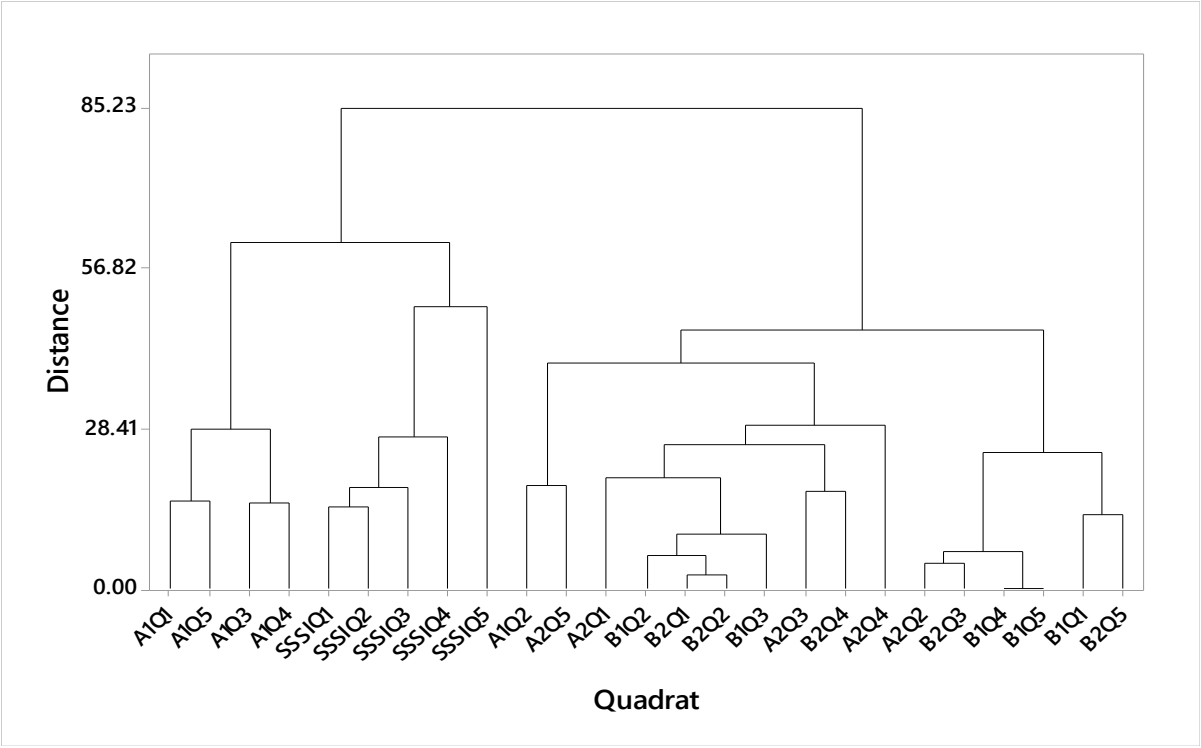


Figure 4.41 Clustering of quadrats in April 2017 (32 months after the breach) using the average linkage method and Euclidean distance. OM is not included because data could not be gathered due to breeding birds on the site.

It is likely that the four A1 quadrats are clustered closer to the SSSI, because they had more coverage of *P. maritima* and less bare ground than quadrats on the other plots, which is like the SSSI. The influence of *P. maritima* on the groupings of these A1 quadrats and the SSSI can be seen in the ordination (Figure 4.42). The absence of OM from the analysis meant that the eigenvalue of DCA1 was relatively low compared to previous and subsequent years, although it was still meaningful. Although OM could not be assessed, this change in eigenvalue provided a good indication that OM was very different in species composition.

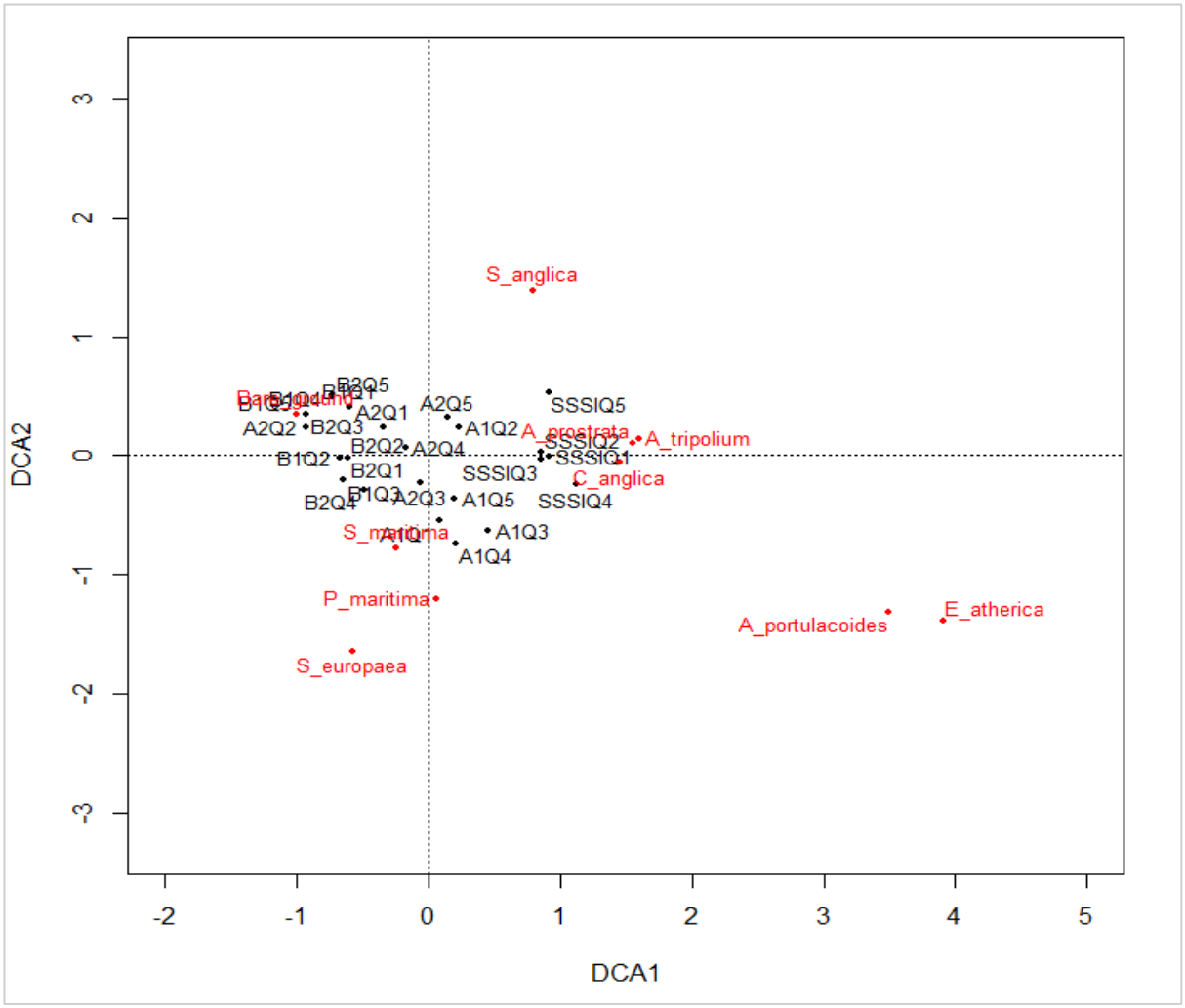


Figure 4.42 Detrended correspondence analysis biplot for April 2017 vegetation data showing groupings of quadrats and the species responsible. Quadrats are in black font and species are in red font. OM is not included because data could not be gathered due to breeding birds on the site. DCA1 had an eigenvalue of 0.5. All other axes had eigenvalues < 0.5.

Cluster analysis conducted on data assessed thirty-six months after the initial breach showed that some of the Site B quadrats were similar to the A1 quadrats that had been close in distance to the SSSI quadrats in April 2017 (A1Q1, A1Q3, A1Q4 and A1Q5). One of the SSSI quadrats was closer to the some of the Site A quadrats than it was to the other SSSI quadrats (Figure 4.43).

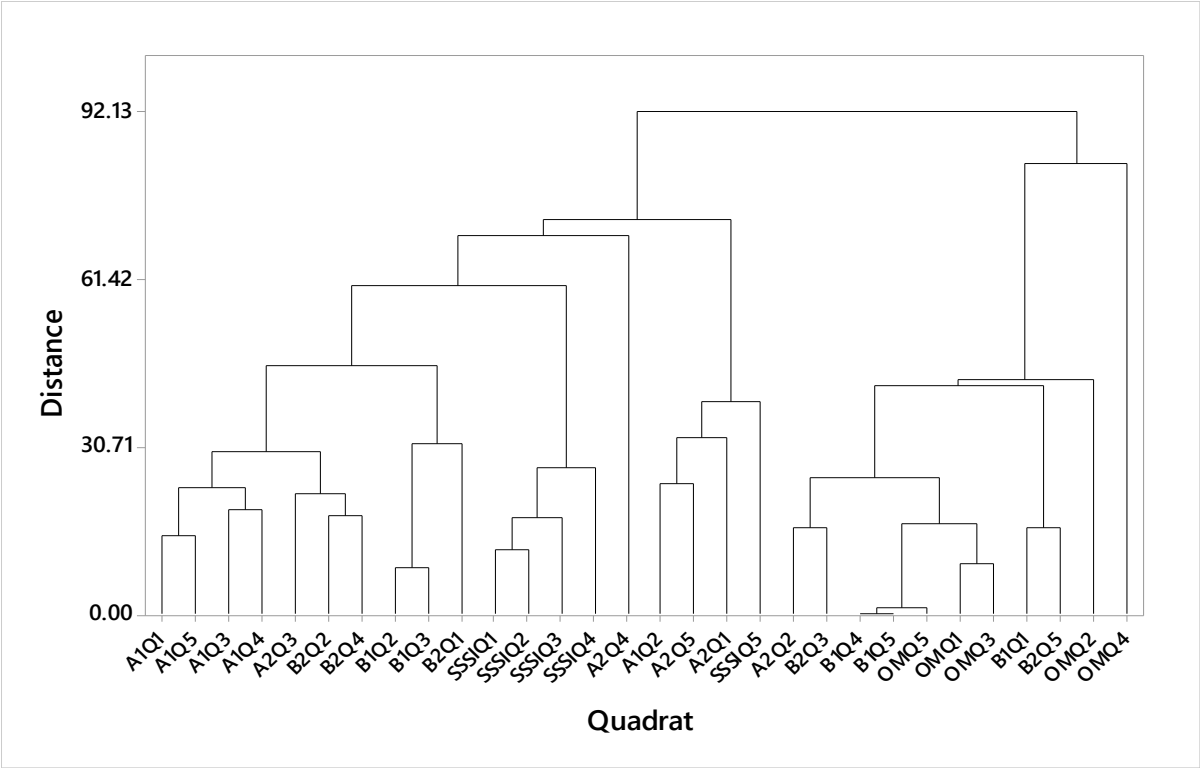


Figure 4.43 Clustering of quadrats in August 2017 (36 months after the breach) using the average linkage method and Euclidean distance.

This grouping was not expected at this stage but appeared to be driven by the high coverage of *S. anglica* in these quadrats; however, it should be noted that DCA2 was not deemed as meaningful on the ordination (Figure 4.44). Some of the Site B quadrats were moving closer to the Site A quadrats because they had a higher coverage of *P. maritima* by this stage. The presence of *A. tripolium* in Site A quadrats influenced the groupings of these in relation to the SSSI. OM quadrats were still very different, and there were still remnants of terrestrial species. However, *S. media* and particularly *S. marina* were relatively abundant in these quadrats by August 2017. Although these species are found on saltmarshes, they were not found in abundance on the other plots or on the SSSI, which is why they are grouped differently on the dendrogram and the DCA ordination.

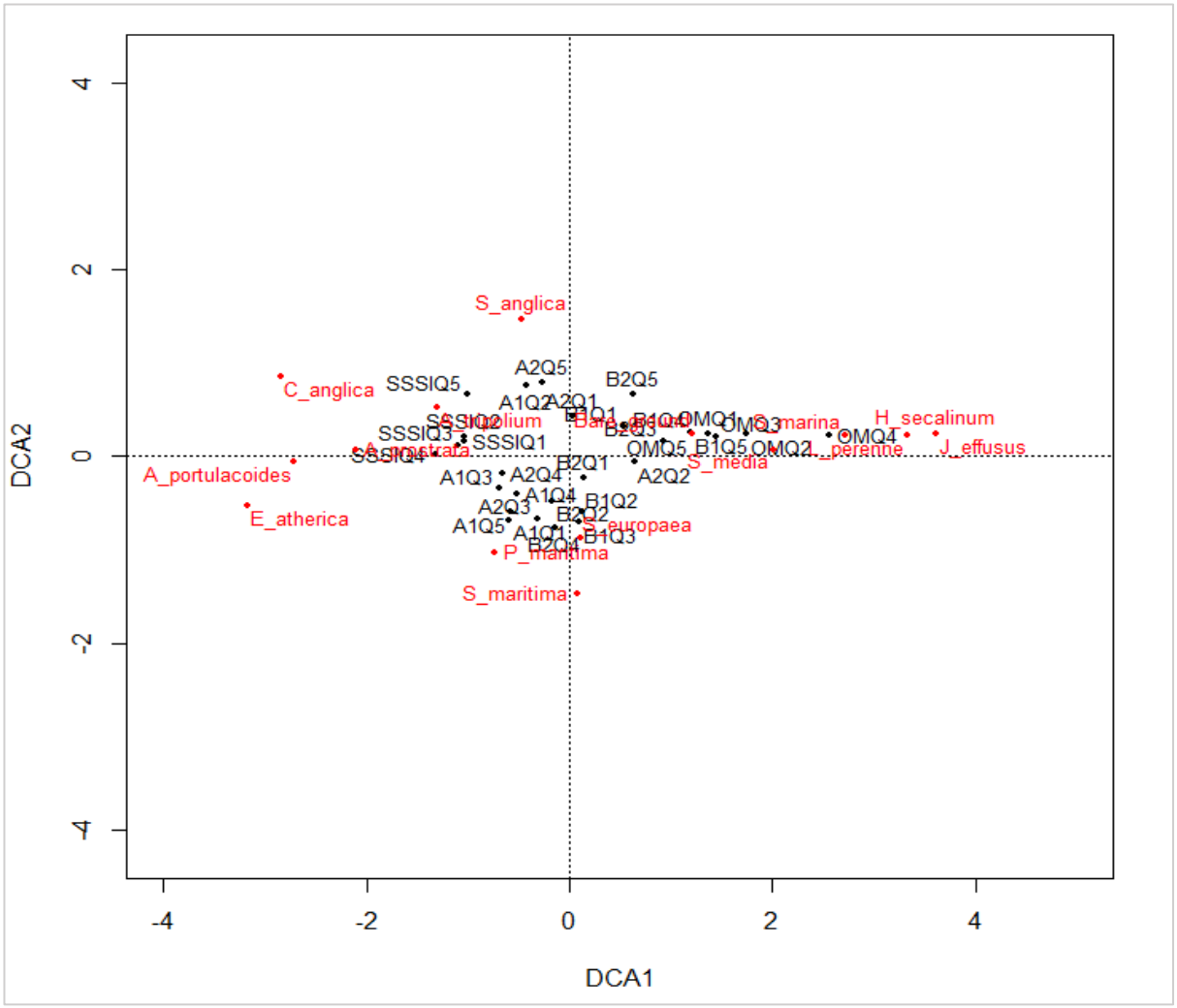


Figure 4.44 Detrended correspondence analysis biplot for August 2017 vegetation data showing groupings of quadrats and the species responsible. Quadrats are in black font and species are in red font. DCA1 had an eigenvalue of 0.6. All other axes had eigenvalues < 0.5.

Clustering and ordination conducted on August 2018 data derived similar results to the previous August, but many of the Site B quadrats were becoming closer to the Site A quadrats and the SSSI. This was at a slower rate, but more characteristic saltmarsh species had started to propagate on Site B by this time. OM was still very different at this stage (Figure 4.45).

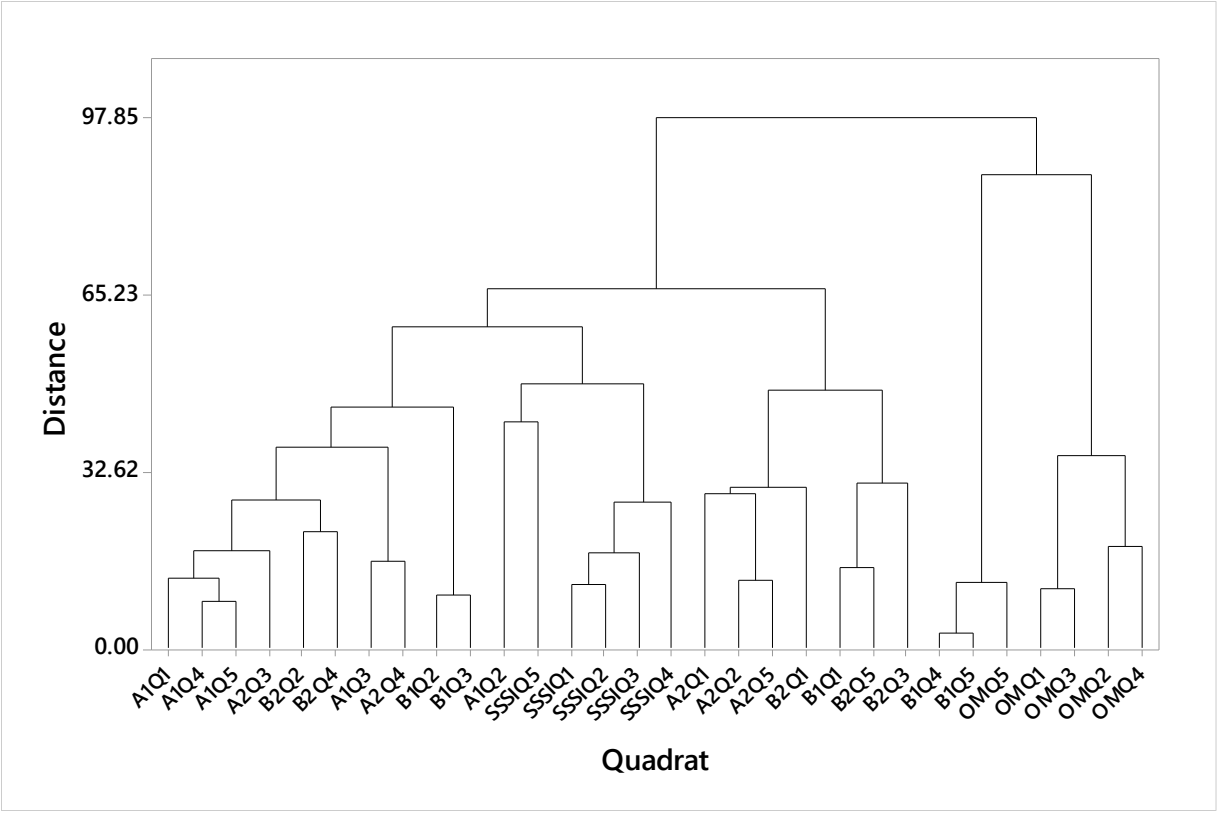


Figure 4.45 Clustering of quadrats in August 2018 (48 months after the breach) using the average linkage method and Euclidean distance.

Ordination shows that most of the A1 and A2 quadrats had similar site scores to the SSSI on DCA1, which was deemed as a meaningful gradient. Only A2Q2 had very different scores by this time. This was due to the amount of bare ground that was still in the quadrat, which can be seen in the ordination plot (Figure 4.46).

By assessing these data in relation to the NVC, it is clear that the Site A plots were the most similar to the SSSI by 2018. A1 plots were the closest in composition to the SSSI, and Site B plots were lagging. Although saltmarsh communities had developed in Site B quadrats by 2018, there were still clear differences from the SSSI and Site A plots in many quadrats, and bare ground persisted. OM was still very different in terms of vegetation coverage. Bare sediment persisted throughout the study on these plots, and they appeared to be dominated by *Spergularia* spp. by 2018.

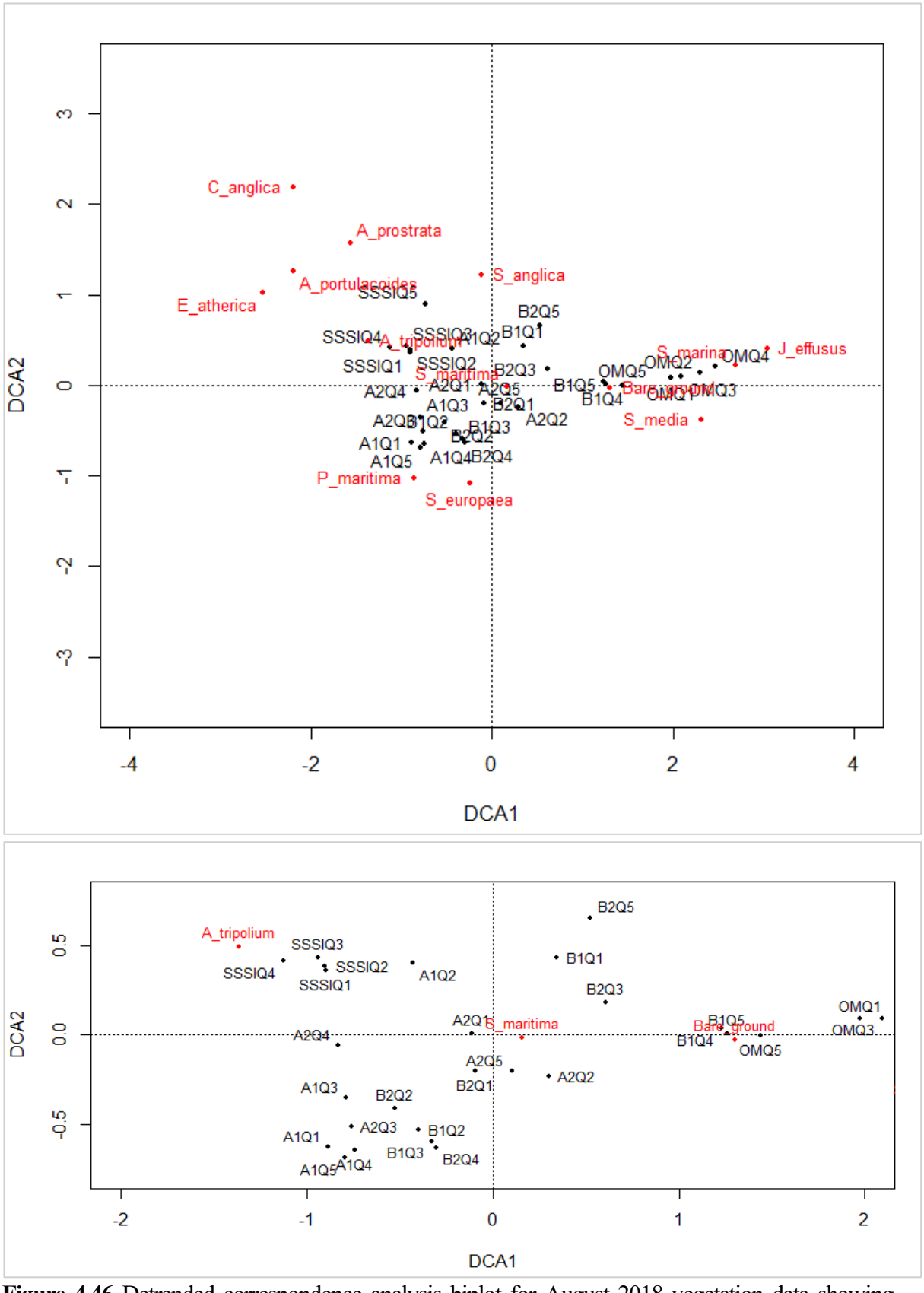


Figure 4.46 Detrended correspondence analysis biplot for August 2018 vegetation data showing groupings of quadrats and the species responsible. Quadrats are in black font and species are in red font. Smaller image shows magnification of the biplot. DCA1 had an eigenvalue of 0.7. All other axes had eigenvalues < 0.5.

4.3.5 Influence of compaction and land height

Although compaction was not tested and quantified until 2018 (see Chapter 3), these data and initial observations suggested that it was likely that the Site B plots and OM were more compacted than the Site A plots. This would have had an impact on the colonisation of species on these sites. Aerial photographs show the effect of this compaction (Figure 4.47-Figure 4.49).

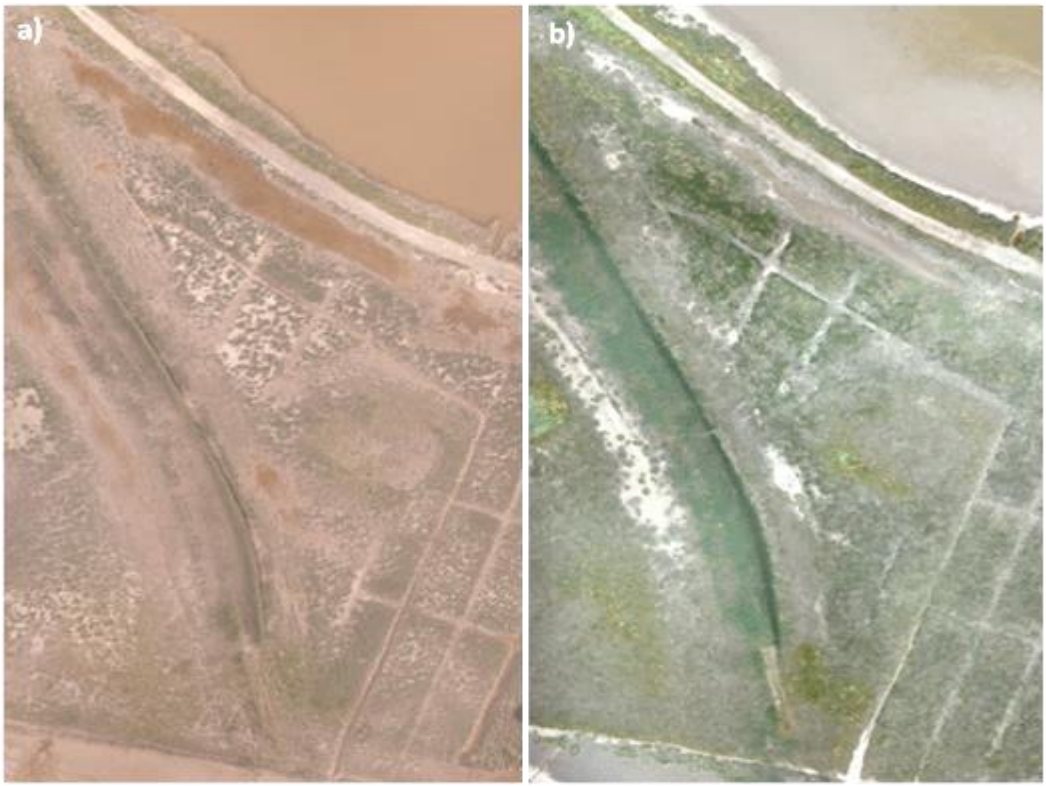


Figure 4.47 Aerial photographs of Site A in a) 2015 and b) 2018 (courtesy of WWT).

Drainage channels remaining from previous land-use can be seen in aerial photographs of Site A (Figure 4.47), but bare ground is not prominent in August 2015 when compared to the other sites. The photographs show how dense the stands were by 2018, and there were only minimal patches of bare ground. Quadrat A2Q2 was in one of these patches.

Site B was very bare in August 2015 and photographs show a large area that contained standing water (Figure 4.48a). This had a negative influence on plant growth, and the legacy of this can be seen in an aerial photograph from 2018 (Figure 4.48b). It is very likely that this area was compacted by machinery during re-profiling of the site. Vegetation had colonised in some of the quadrats around this area, but it is likely that the standing water did slow development.

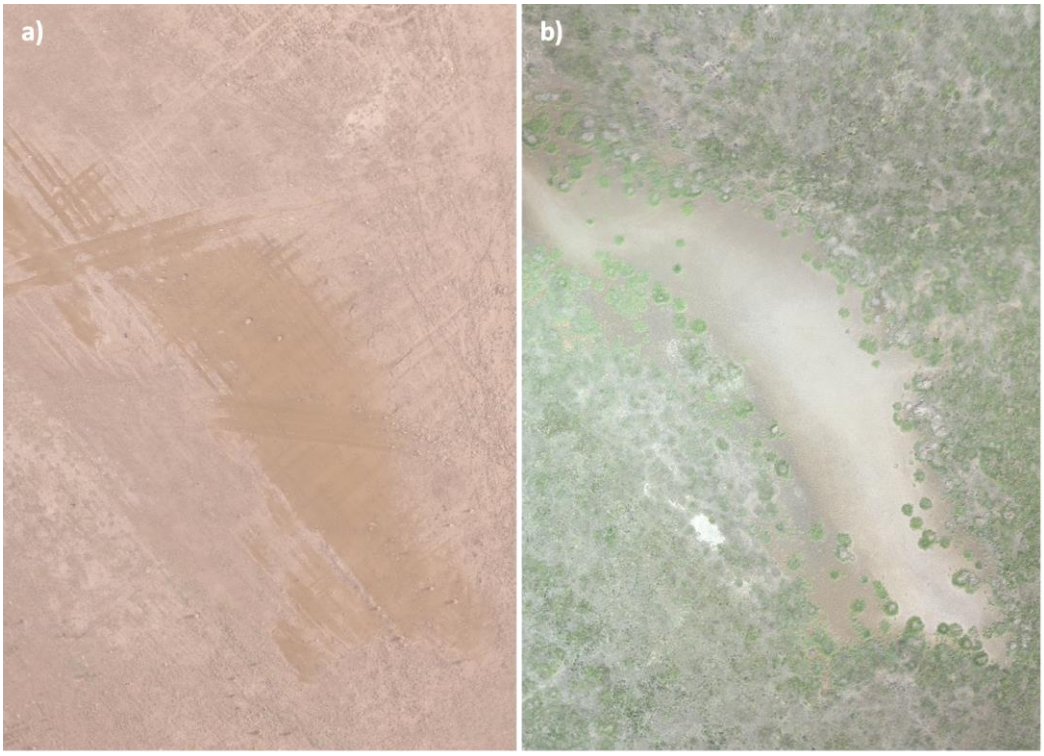


Figure 4.48 Aerial photographs of Site B in a) 2015 and b) 2018 (courtesy of WWT). The standing water shown in a) covers an area of approximately 1700 m^2 (measured on Google Earth Pro) and the study plots were partially in this area. The same area is shown in b) and reveals that vegetation coverage was still minimal in 2018.

Aerial photographs of the OM study site show that vegetation coverage was very low in August 2015 (Figure 4.49a). Data and observations indicate that this area was also compacted, and water was often held in this area to supplement scrapes, which is part of the ongoing management of the site, particularly for breeding birds. Sheep and cattle had also grazed this site during the study, which is part of the agreed management of the scheme. A subsequent aerial photograph taken in 2018 (Figure 4.49b) shows that vegetation coverage was still very limited, although there was some growth. Ground truthing showing that this mainly consisted of *Spergularia* spp. (see sections 4.3.2 – 4.3.4).

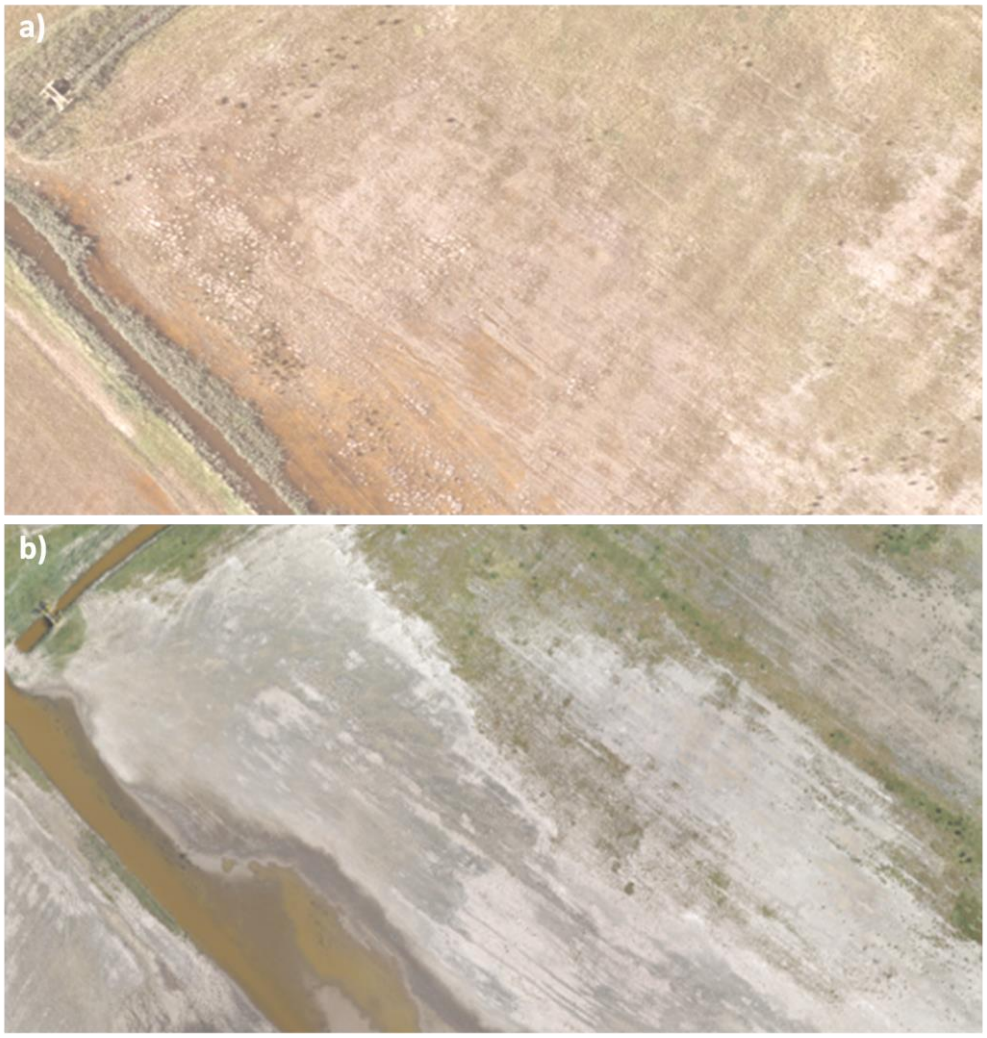


Figure 4.49 Aerial photographs of OM in a) 2015 and b) 2018 (courtesy of WWT).

Bare ground (%) was tested against height of land (metres AOD) using a Spearman rho correlation (Figure 4.50), chosen because data were not normally distributed. Elevation data from 2016 and 2018 were combined for this analysis (see Chapter 3). It was found that there was a significant negative correlation between land height and % bare ground in A1 and A2 quadrats combined (r_s = - 0.54, p = 0.015; Figure 4.50a) and in B1 and B2 quadrats combined (r_s = - 0.79, p < 0.001; Figure 4.50b). There was not a correlation between land height and % bare ground in OM (r_s = - 0.01, p = 0.987; Figure 4.50c), but there was a correlation when all quadrats were included in the analysis (r_s = - 0.44, p = 0.001; Figure 4.50d).

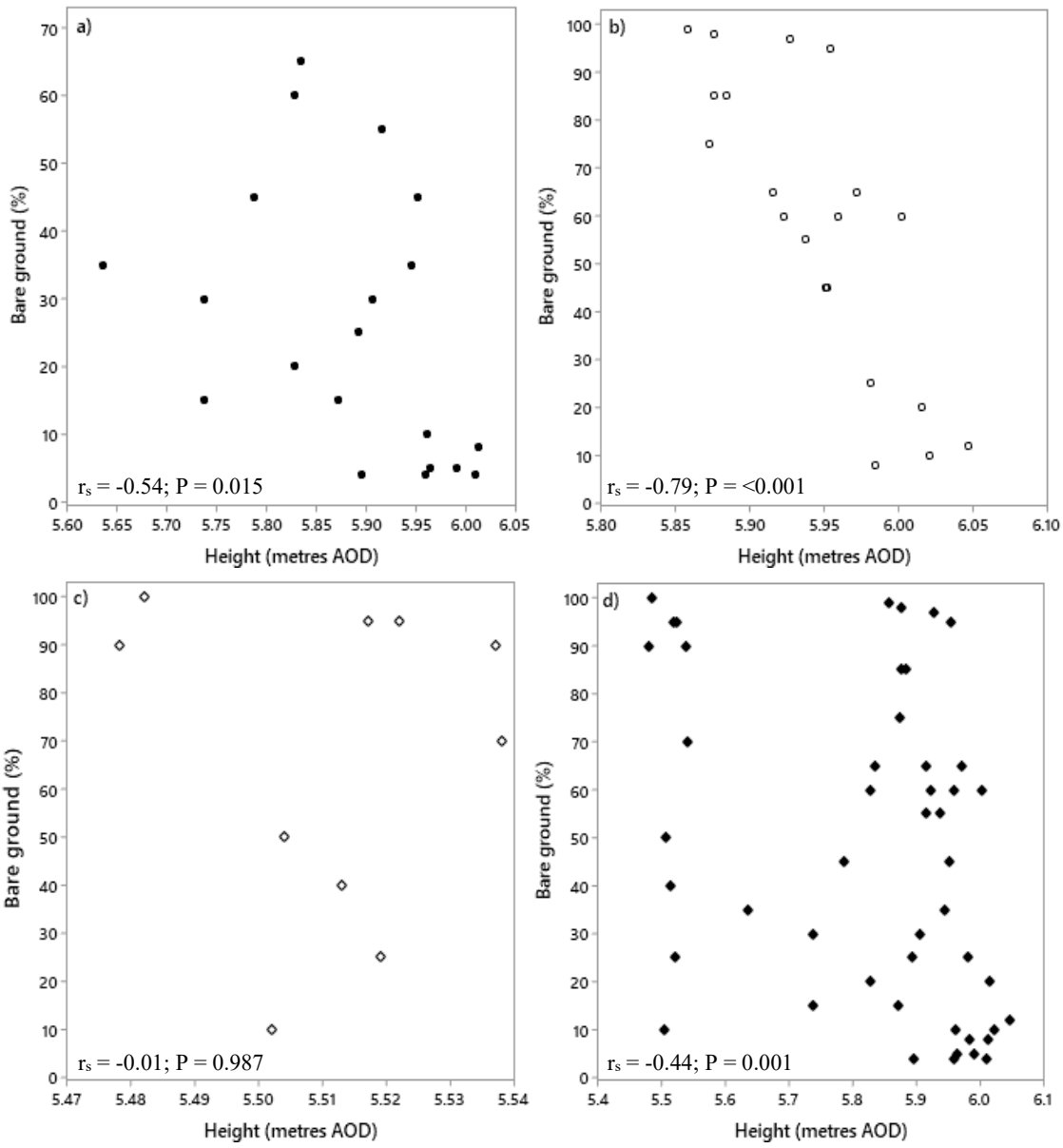


Figure 4.50 Spearman rho correlation between land height (metres AOD) and % bare ground coverage. Data derived from combined 2016 and 2018 quadrat heights and % bare ground in a) A1 and A2 (n = 20); b) B1 and B2 (n = 20); c) OM (n=10); and d) A1, A2, B1, B2 and OM (n = 50).

4.4 Discussion

The aim of this chapter was to summarise vegetation species composition on Steart Marsh in comparison to the SSSI saltmarsh (target state) and the managed OM site, and determine trends of community changes in relation to the target state throughout the study period.

The key findings within this chapter are that existing vegetation died back quickly on Steart Marsh plots after the breach, but halophytic vegetation species colonised Site A plots more rapidly than Site B and Otterhampton Marsh, and vegetation community composition was more similar to the SSSI target state on Site A plots than to all other plots

by the end of the study. Otterhampton Marsh was different to all other plots throughout the study and changed more slowly. There were remnants of glycophytic species for some time on OM and species richness was lowest on this site. Few halophytes colonised the OM plot throughout the study period. Bare ground was prevalent on Site B and OM throughout the study. Some Site A quadrats had bare ground during the study, but dense swards of halophytic vegetation were found on these plots by the end of the study, and this was more similar to species composition on the SSSI than to the plots on the other study sites (Site B and OM).

The composition of vegetation communities changed since the breach of the existing sea defences in September 2014. Grass species, such as *A. stolonifera*, and *H. lanatus,* wildflowers such as *T. repens*, and ruderals such as *C. vulgare* and *C. arvense* were replaced rapidly by pioneer coastal halophytic species, but colonisation by saltmarsh flora was not rapid on all plots. Initially, there was a large amount of bare ground across all plots, which could be due to the tidal regime burying the existing terrestrial species in sediment, although the effects of sedimentation in the plots would need to be investigated to confirm this. The prevalence of bare ground in the initial stages could have been due to unsuitable starting conditions or the lack of available propagules. It is likely that changes in pH, salinity and hydrology caused die-back of the existing vegetation and allowed for the colonisation of pioneer saltmarsh species. *A. prostrata* was one of the first species to colonise in 2015, with the greater coverage being on the site that was pasture before the breach (Site A). The species may have had a greater coverage on Site A because it was less compacted than Site B and OM, and standing water was not present on this site. This annual herbaceous orache species was dominant and had little competition at this stage, and it was also present on the SSSI (Mean = 24% cover), possibly explaining why it was one of the first species to establish on Steart Marsh. *S. maritima*, another annual species, also present on the SSSI, was an early coloniser and was found consistently in plots with *A. prostrata* on Site A. This species was also an early coloniser of the Blackwater estuary MR schemes (Tollesbury and Abbots Hall) in the South East of England (Wolters *et al.*, 2008; Hughes, Fletcher and Hardy, 2009).

The SSSI saltmarsh most closely resembled an SM12 community, which is dominated by *A. tripolium*. This is a biannual species that forms large stands. There were also matches with an SM10 community (transitional low-marsh vegetation), mainly due to the presence of *P. maritima* and *S. maritima*, but *Salicornia*, another constant of this NVC community, was not found in the SSSI quadrats. This may be because it has been displaced by *S. anglica*, which can grow at the expense of *Salicornia* at similar elevations (Adam,

1990). SM6 was another match because of the abundance of *S. anglica* and the presence of other SM6 species, including *P. maritima* and *A. tripolium*. According to Rodwell (2000), *A. prostrata* and *S. maritima* are conspicuous on a few SM6 sites and form distinctive sub-communities, which could have been the case on the SSSI site. However, the presence and relative abundance of *A. tripolium* on the study plot indicated that SM12 was a reliable match for the SSSI, especially because *S. anglica* and *P. maritima* are frequently associated with *A. tripolium* in this NVC community (Rodwell, 2000). There is thus an expectation that the MR sites might develop into SM12 communities.

S. anglica and *P. maritima* started to colonise Both A and B sites in August 2015, but there was initially low coverage of these species. At first, they formed small clumps and did not form substantial swards until 2017. *Spartina* species usually colonise bare sediments on marshes, including mud-flats (Huckle, Potter and Marrs, 2000), and there is evidence that *S. anglica* did propagate on bare sediment on Steart Marsh. This has some benefit to the establishment of the marsh community at Steart because *Spartina* grasses facilitate accretion (Doody, 2008). However, the spread of *S. anglica* on the MR site may also reduce feeding areas for birds, such as waders, especially because *Spartina* stands can limit their ability to retrieve invertebrates from the sediment (Hammond and Cooper, 2003). The spread of *S. anglica* is likely to continue on Steart in the short-term and may have implications for the future role of the site for some wader species, whose presence is one of the objectives of the MR scheme. Management of *Spartina* is potentially expensive and difficult in an estuarine environment, because it may need to be removed mechanically (Prieto *et al.*, 2011; Strong and Ayres, 2013). Hammond and Cooper (2003) suggest that herbicide is a cost-effective method, but it requires multiple treatments, and other marsh species are likely to be affected. Doody (2008) recommends that management of *S. anglica* should be carried out on a case-by-case basis, because of the benefits it can provide for specific saltmarshes, and there is evidence that *Spartina* has declined naturally in Bridgwater Bay due to wave action (Morley, 1973; Doody, 2008). Whether this natural decline will eventually occur on Steart Marsh will require further monitoring and may not be discernible for many years.

P. maritima was also abundant on Site A and B plots by August 2017, and especially by August 2018. It appears to commonly colonise MR sites, often becoming dominant (Burden *et al.*, 2013; Sullivan *et al.*, 2018). There is evidence to suggest that this species also plays a key role in sediment stabilisation and accretion (Langlois, Bonis and Bouzillé, 2001; Langlois, Bonis and Bouzillé, 2003; Reef *et al.*, 2017), so there are benefits from its rapid colonisation and proliferation on Steart Marsh. Coverage in some quadrats on Site A were

higher than on the SSSI. This may be beneficial for wild fowl such as wigeon (*Mareca penelope*) which tend to graze *Puccinellia* swards on marshes (Adam, 1981). It was likely that the species would colonise and spread rapidly because of its tolerance of waterlogging and salt (Cooper, 1982), and it has the ability to produce stolons that help it extend over large areas (Langlois, Bonis and Bouzillé, 2001).

A. tripolium stands were increasing each year on the Site A plots in particular, but the marshes were seasonally grazed by 48 Longhorn cattle from May 2017. These cattle were removed from the site in October 2017 and put back out to graze in the following April. After grazing commenced there were signs of poaching by the cattle in the study areas and stands of *A. tripolium* had been eaten, which potentially reduced the coverage of this species. The cessation of grazing by livestock has been shown to instigate an increase in *A. tripolium* on saltmarshes (Schröder, Kiehl and Stock, 2002), so this may have an influence on a direct comparison between the grazed MR, OM and the SSSI plot. The SSSI had not been grazed by livestock during the study period, although grazing by wild fauna is likely to have occurred on all sites and the SSSI.

Bare ground was evident throughout the study on all MR plots, and this persisted in some quadrats, which had very little vegetation coverage throughout the duration of the study. It was expected that some areas would remain unvegetated because this has occurred on other MR sites after 4-7 years (Davy *et al.*, 2011; Mossman *et al.*, 2012; Brooks *et al.*, 2015). There was a negative correlation between land height and bare ground in this study, with lower land tending to have a greater coverage of bare sediment. This was not the case on OM, but land height was very similar in all quadrats, whereas there was more variation on the other plots. According to Ivajnšič, Šajna and Kaligarič (2016), even very small changes in elevation can influence plant zonation on saltmarshes, so it is not surprising that bare ground persisted in areas that were lower in the tidal frame, even though they were originally chosen to be within 0.5 metres in height AOD of one another. The lower quadrats were likely to have been submerged for longer periods during tidal inundation, which can impede plant growth due to a lack of oxygen in the sediment (Masselink *et al.*, 2017).

By August 2017, seven characteristic saltmarsh species were found on Site A plots (*A. prostrata*, *A. tripolium, P. maritima, S. anglica, S. europaea, S. maritima* and *S. media*). *C. anglica* had also colonised and was found on Site A plots during the study, but this was not found in the quadrats in August 2017, and there was no longer evidence of an algal mat that was observed in 2016. In comparison, there were five species found on Site B plots in August 2017 (*A. tripolium, P. maritima, S. anglica, S. europaea and S. maritima*), and three

on OM (*S. marina, S. media and S. maritima*). Eight species were observed on the SSSI plot in August 2016 (A. *portulacoides, A. prostrata, A. tripolium, C. anglica, E. atherica, P. maritima, S. anglica and S. maritima*).

Most of the halophytic species found on the Steart Marsh MR were present on the SSSI saltmarsh sample plot, which suggests that the mature marsh was acting as a seedbank. This reinforces the view by Wolters, Garbutt and Bakker (2005) that successful restoration of saltmarsh flora is facilitated by proximity to a local source of saltmarsh species. The quadrats on the mature SSSI site were at approximately the same height (AOD) as the starting position of the MR marsh quadrats, so it was unsurprising that similar vegetation started to colonise. The MR plots were also near to the breach, so it is reasonable to suggest that seeds were more readily deposited at this stage, which left less available as the tidal water travelled across the marsh. Anecdotal evidence suggested that this was the case, because areas further from the breach had noticeably less vegetation cover, especially in the first-year post-breach. There was also the possibility that seeds were being distributed by other abiotic and biotic processes, such as by wind dispersal or by fauna, but it is assumed that tidal inundation is the primary means for seed dispersal on an MR site (Wolters, Garbutt and Bakker, 2005), so it was therefore likely to be the main driver for dispersal at the Steart MR scheme.

Data show that the site that was previously pasture (Site A) started to tend towards those communities seen in the adjacent SSSI saltmarsh more rapidly than the other sites (Site B plots and OM). The NVC matches on the Site A plots were similar to those on the SSSI, with the highest match being with an SM10 community in August 2017, although SM12 was also a high match. SM10 is dominated by *P. maritima, Salicornia spp.* and *S. maritma*, which are common in transitional low marsh vegetation, so this was not unexpected on a new site. According to Rodwell (2000), this type of community is found on clays with a pH range of 7.0-8.0 which draws a parallel with the pH of the plots (see Chapter 3). The impact of the livestock on the plots may have had an influence on the community match, especially because of the reduction of *A. tripolium*. However, by August 2018 SM12 was a high match, and ordination showed that most Site A quadrats (on A1 in particular) were very similar in species composition to the SSSI.

The site that had been in an arable crop rotation pre-breach (Site B) was changing at a slower rate than the Site A plots, possibly due to compaction caused by previous land use or the use of heavy machinery during construction, which may have facilitated the accumulation of standing water throughout each year. If MR sites do not drain, high levels

of submergence can have a negative impact on plant establishment and growth (Masselink *et al.*, 2017), so it is likely that the standing water impeded growth due to anaerobic conditions. The heavily compacted soil would have also affected the ability of plants to root. There was more growth of saltmarsh species observed on either side of the plots on this site, and these areas were not subjected to the same level of submergence from the standing water. *A. tripolium* was not present in the Site B quadrats until 2017, and this only consisted of a few individuals in B2, although more stands were seen around and between these quadrats. This species was far less abundant on these plots compared to the moderate to high coverage on Site A plots. Seed dispersal of this species is by water and also wind, but it has little vegetative spread once deposited (Fitter and Peat, 1994), which may have limited its ability to propagate on the Site B plots. *S. anglica* did colonise these plots, with some stands reaching approximately 70cm in height by August 2017. This was not unexpected because this species tends to form communities in pans, as well as on other bare sediment (Adam, 1990).

P. maritima had grown substantially in 70% of the Site B quadrats by 2018 (between 12-65% coverage). The site had accreted by 2018 and appeared to be levelling at marginally below 6 metres AOD. This would have reduced submergence time, which may be a reason why *P. maritima* had proliferated by 2018. However, the species appears to grow well in sediment with low redox potential (Davy *et al.*, 2011), which often occurs in submerged sediment (Tokarz and Urban, 2015). It is likely that redox potential was low on the Site B plots due to the level of standing water on the site throughout the study. *S. europaea* colonised the Site B plots and had relatively high coverage in two of the B1 quadrats. This is reported as a common coloniser on other MR schemes (e.g. Mossman *et al.* 2012; Brooks *et al.*, 2015) and tends to be able to withstand sediments with low redox potential (Davy *et al.*, 2011), which is why it may have colonised on the Site B plots. In the future, redox potential should be quantified on these specific plots to assess if it is having an influence on species composition.

Otterhampton Marsh developed very slowly compared to the plots on Steart Marsh. The plot mainly consisted of bare ground and there were still remnants of terrestrial species in 2016. It was evident that the plot was poached heavily by livestock, with 40-50 sheep grazing OM in 2015, and 9 Dexter cattle grazing the site in 2017. This, along with the brackish, rather than saline, influence may have impacted the colonisation of characteristic saltmarsh species. Compaction may have also been an issue on OM and it was not always left to a natural tidal regime, as water held in this area to maintain water levels in nearby

scrapes may have had a similar effect to the standing water on the Site B plots. However, OM is further away from a natural saltmarsh seedbank than the plots on Steart Marsh, and it is slightly lower in elevation. This means that the delivery of seeds may have been at a slower rate and immersion times may have been longer, also leading to anaerobic conditions and reduced plant growth. Nevertheless, by August 2018 *Spergularia* spp. had colonised in large numbers. SM23 was reported in the NVC analysis, although the matches were low. This community has *S. marina* as a constant, but at this stage there was a lack of other species that form this community such as *P. maritima* and reflexed saltmarsh grass (*Puccinellia distans*). However, the latter was not found on the mature marsh or Steart Marsh either, and it is not reported in this part of the Severn Estuary (Fitter and Peat, 1994).

The Site A plots were closer to the SSSI seedbank than the other plots, which may have had an effect on the speed of colonisation of similar species in these quadrats, and the high coverage of vegetation on the site pre-breach may have helped to trap sediment in the initial stages, allowing for quicker accretion over the study; however, it was still lower in elevation than the Site B plots by 2018. The lack of standing water on Site A and lower levels of compaction are likely to be the primary reasons why the site has developed at a faster rate of change than other studied plots.

The results of this study clearly indicate that vegetation establishment on Steart Marsh is not uniform and likely depends on tidal regime, distance from propagule sources, the original and subsequent soil condition and current management on the site.

CHAPTER 5: INVERTEBRATE COLONISATION

5.1 Introduction

Changes of the vegetation communities and soil characteristics on the Managed Realignment site are expected to have an impact on the invertebrate fauna that colonises the site post-breach. Conversely, invertebrate fauna play a key role in influencing soil dynamics and plant productivity (Boorman, 2003; Silliman and Bortolus, 2003), and they are essential components in food webs. Research based on invertebrate colonisation on MR schemes is limited and it has mainly concentrated on benthic invertebrates (see Evans *et al.*, 1998; Atkinson *et al.*, 2004; Mazik *et al.*, 2007), so it is important to gather data on a range of different invertebrate groups (ground-dwelling, plan-dwelling and soil-dwelling) to determine how rapidly these groups colonise.

The aim of this chapter is to report and summarise ground-dwelling, flying and soil dwelling/benthic invertebrate species colonisation and composition on Steart Marsh in comparison to the managed OM site and the SSSI saltmarsh (target state), and determine trends of change in relation to the target state throughout the study period. Invertebrate family group composition is reported each year from 2014 (pre-breach) to August 2017. Detrended Correspondence Analysis is used to show how invertebrate composition in study plots (A1, A2, B1, and B2) compare to the target state and the managed OM site each year of the study. This will provide information on the rate of invertebrate colonisation and change in family group composition in the different study plots on Steart Marsh and Otterhampton Marsh. An inventory of family groups is reported each year by quadrat on each study plot and the results from cluster analysis and ordination techniques are reported and discussed in this chapter. Key invertebrates found across sample plots are identified to species and implications are discussed.

It was expected that the initial invertebrates pre-breach would be terrestrial and characteristic of agricultural land on Steart Marsh and Otterhampton Marsh, but the abundance and richness of these would drop quickly following the breach and change with invasion by families of halophile coastal invertebrates colonising the sites. Many invertebrate species on saltmarshes are phytophagous (Adam, 1990), so the plants that have colonised will have a direct influence on the invertebrate species that will be found. It is expected that the specific species of plant and their abundance would have an influence on the specific invertebrates on the Steart Marsh plots and OM, with lower colonisation of phytophagous invertebrates on plots with extensive bare ground because of the lack of cover

and plants to feed and reproduce on. The potential compaction of the soil (see Chapter 3) may also have limited the movement of, and colonisation by, soil-dwelling invertebrates in these plots, although it was expected that individuals may have been found in the estuarine sediment that arrived during tidal emersion, which was the case at the Tollesbury MR scheme (Garbutt *et al.*, 2006).

5.2 Methods

Invertebrates were sampled using Common Standards Protocols, whereby 5 pitfall traps were placed in each permanent quadrat for 9 days once a year (late summer). The traps were placed in each corner of a quadrat and one in the middle. The 9-day period was chosen because of the small window between high spring tides. Each pitfall trap contained saline solution and scentless washing-up liquid to break the surface tension. The five pitfall traps from each quadrat were collected on the 9th day and combined. They were poured into plastic containers, which were labelled with site name, quadrat number and date. Later, the invertebrates in each container were rinsed, stored in water for 20 minutes and rinsed again to remove the detergent. The invertebrates were carefully placed in sample tubes that were labelled with site name, quadrat number and date. The invertebrates were preserved in 70% clear industrial methylated spirits (IMS).

Sweep net sampling was carried out once per year (late summer). Each quadrat was swept with a sweep net (including up to 1m outside of the quadrat). A pooter was used to collect invertebrates found in the net. Each pooter tube was labelled with site name, quadrat number and date. The invertebrates were later refrigerated to slow down their movement and then preserved in 70% clear IMS.

Soil-dwelling invertebrates were recorded in soil cores from each quadrat. Soil cores were obtained once per year (August). Three 20cm cores were taken from each quadrat. Each core was split into two sections (C1: top 10cm; C2: 10-20cm depth). The three cores from each section were combined and stored in a refrigerator in sample bags for later analysis. Each soil sample was weighed and decanted into sample trays. Each sample was broken up by hand and any invertebrates found in the soil were collected and stored in labelled sample tubes with the quadrat number, depth (C1 or C2) and date written on the labels. 70% clear IMS was used to preserve the invertebrates. Disposable gloves were used when the soil was broken up to ensure that it did not get contaminated, and the soil was re-bagged and placed in the refrigerator. These measures were necessary because the sample was also used to determine soil characteristics (see Chapter 3).

Invertebrates in each of the tubes were sorted into groups and placed in labelled tubes in IMS. Later, each tube was decanted into petri dishes and placed under a dissection microscope. Forceps were used to manipulate the invertebrates and invertebrate guides and taxonomic keys were utilised to aid identification to family level.

5.2.1 Statistical analysis

Minitab 18 was used for initial descriptive data analysis to show invertebrate family numbers in plot quadrats, and how these changed each year of the study. This statistical package was also used for Spearman Rank correlations to assess the relationship between plants and associated phytophagous invertebrates, and for cluster analysis to show similarities of invertebrate assemblages in quadrats within study plots each year. DCA ordination and Procrustes error plots were carried out on the Vegan package (Oksanen *et al.*, 2017) to show how quadrats were ordinated in relation to invertebrate families and how quadrat groupings changed from 2014-2017. R version 3.5 was used to conduct Kruskal-Wallis tests and subsequent post-hoc tests using the PMCMRplus package (Pohlert, 2018) to determine if there were differences in a phytophagous dipteran species each year.

5.3 Results

In this study a total of 6,337 invertebrate specimens were caught in pitfall traps across the study sites (A1-OM) from August 2014-August 2017. Additionally, circa. 4712 invertebrates were caught on the SSSI in 2016 (numbers of an amphipod species were estimated due to substantial quantities). A total of 1,634 specimens were caught in sweep nets across all sites (A1-OM) from August 2014-August 2017, and an additional 242 were caught on the SSSI plot in August 2016. Specimens were examined to family level across all plots. Forty-nine families were identified in total in pitfall traps and one was identified to order level. A total of fifty-six families were identified from specimens caught in sweep nets.

5.3.1 Ground-dwelling invertebrates

5.3.1.1 SSSI target state

Amphipods in the family Talitridae dominated the saltmarsh invertebrate fauna in pitfall traps on the SSSI (August 2016). Data were extrapolated by filling three tubes with amphipods to the same level, counting the amphipods in three tubes, averaging the quantities, filling further tubes to the same level, and multiplying the number of tubes by the average quantity. It was discovered that there was c.4400 amphipods in 25 pitfall traps on the SSSI. Aranaea were dominated by Lycosidae (85 individuals) in the genus *Pardosa*.

Tetragnathidae spiders in the genus *Pachygnatha* were also found on the marsh (12 individuals) (Table 5.1).

Table 5.1 Invertebrate families found in pitfall traps on the SSSI study site in August 2016.

Class	Order	Family	SSSI QTY
Arachnida	Araneae	Linyphiidae	2
		Lycosidae (genus *Pardosa*)	85
		Tetragnathidae (genus *Pachygnatha*)	12
Gastropoda	Ellobiida	Ellobiidae	1
Insecta	Coleoptera	Carabidae	17
		Latridiidae	3
		Staphylinidae	1
	Diptera	Chloropidae	3
		Phoridae	1
		Scathophagidae	18
		Sphaeroceridae	98
	Hemiptera	Cicadellidae (genus *Anoscopus*)	67
		Delphacidae	2
	Hymenoptera	Braconidae	1
Malacostraca	Amphipod	Talitridae (genus *Orchestia*)	c.4400
	Isopoda	Sphaeromatidae (genus Lekanesphaera)	1

Carabid beetles were found in low numbers in pitfall traps (17) and only one rove beetle (Staphylinidae) was recorded. There were 63 Cicadellidae (leafhopper) specimens found in the traps, all in the genus *Anoscopus*. One isopod of family Sphaeromatidae was recorded in SSSI Q3 (Figure 5.1a). The species was identified *as Lekanesphaera rugicauda* (Leach, 1814).

Numbers of Talitridae were not evenly spread between quadrats in the SSSI plot, with a disproportionately high amount in quadrats SSSI Q1 and SSSI Q2, although they numbered in the hundreds in the other three quadrats (Figure 5.1b).

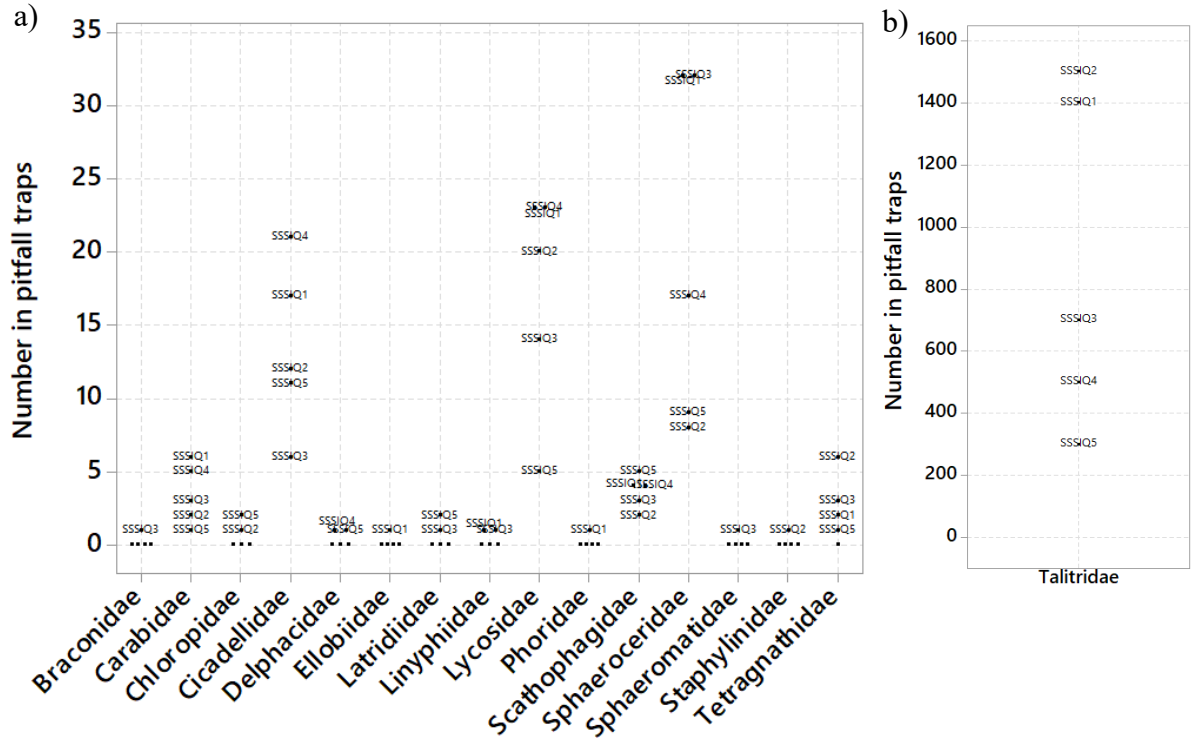


Figure 5.1 Number of invertebrates at family level caught in pitfall traps on the SSSI saltmarsh (August 2016; Q1-Q5; n = 5); Individual data points show the number of invertebrates within a family found in each quadrat; a) invertebrate families caught in pitfall traps in August 2016 apart from Talitridae; b) Talitridae caught in pitfall traps in August 2016 on the SSSI.

5.3.1.2 Steart Marsh and Otterhampton Marsh 2014 (pre-breach)

In August 2014, 2,780 individual invertebrates were caught in pitfall traps across all study plots (A1-OM; excluding the SSSI) within 24 identified families and one order (Pulmonate slugs were identified to order level). Invertebrate types were not evenly spread across each study plot (Table 5.2).

Table 5.2 Total invertebrate quantities and family groups caught in pitfall traps over a 9-day period on each study plot in August 2014. SSSI (collected August 2016) is included for comparison. *True of all plots apart from SSSI, which are all recorded at family level.

	A1	A2	B1	B2	OM	SSSI
Total invertebrate quantity	620	428	693	775	264	c.4712
Family group quantity (includes 1 at order level*)	15	14	12	13	18	16

The most prevalent ground-dwelling invertebrate orders caught in pitfall traps were Araneae and Coleoptera, both with numbers of > 100 on all plots apart from beetles on OM

(64 individuals). Wolf-spiders (Lycosidae) were the most numerous arachnids on plot A1 (167) and A2 (170), and they were relatively numerous on B1 (138) and B2 (140). There were fewer on OM (67). Money spiders (Linyphiidae) were found on all plots but were particularly abundant on B1 (306) and B2 (348). The Tetragnathidae found on these plots were of the genus *Pachygnatha*, which are ground-dwelling long-jawed spiders. There were similar numbers of these arachnids on all plots ranging from 9-38 individuals. Harvestman (Opiliones) were present on all plots, but they were most abundant on plot A1 (54) and numbers were lowest on OM (2). One individual of the family Thomisidae was found on B2, but these spiders were absent from all other plots (Table 5.3).

Table 5.3 Invertebrate families found in pitfall traps on Steart Marsh study plots (A1-B2) and Otterhampton Marsh in August 2014.

Class	Order	Family (Subfamily)	A1 QTY	A2 QTY	B1 QTY	B2 QTY	OM QTY
Arachnida	Araneae	Linyphiidae	49	66	306	357	53
		Lycosidae	167	170	139	140	67
		Tetragnathidae	38	11	9	23	19
		Thomisidae	0	0	0	1	0
	Opiliones	Phalangiidae	54	13	15	17	2
Chilopoda	Lithobiomorpha	Henicopidae	0	0	0	0	1
		Lithobiidae	0	1	0	0	0
Gastropoda	Pulmonata	Unidentified*	48	32	60	70	35
Insecta	Coleoptera	Carabidae	152	100	152	165	62
		Staphylinidae	20	20	0	2	1
	Diptera	Calliphoridae	0	3	2	5	2
		Chloropidae	0	0	1	0	4
		Dolichopodidae	0	0	1	1	1
		Opomyzidae	0	0	2	0	1
		Phoridae	0	1	5	1	8
		Scathophagidae	0	0	0	0	2
		Sciaridae	1	0	0	0	1
		Sphaeroceridae	1	0	0	1	3
	Hemiptera	Cicadellidae	1	2	0	1	0
		Delphacidae	0	0	0	0	1
		Miridae	1	2	0	0	0
	Hymenoptera	Formicidae (Forminae)	48	4	0	0	1
		Formicidae (Myrmicinae)	40	0	0	0	0
		Ichneumonidae	1	0	2	0	0
Malacostraca	Isopoda	Armadillidiidae	2	0	0	0	0
		Philosciidae	5	19	0	0	0

Ground beetles (carabids) were the most abundant beetle family found on all plots, but A1 and A2 had the highest number of rove beetles (Staphylinidae), and a small number of click-beetles (Elateridae) were found on A1 but absent from the other plots. Beetle larvae were only found on A1 and A2.

Of the Hymenoptera, two subfamilies of ants were identified, Formicinae and Myrmicinae. These were most abundant on A1, and sparse or absent from other plots, suggesting that there may have been nests on the A1 plot, although they were only abundant in A1Q4 (Figure 5.2a). Land-dwelling slugs (Pulmonata) were present on each plot, with the lowest numbers on A2 (32) and OM (35) and the highest on B2 (70). A1 and B1 had similar numbers of Pulmonata (50 and 60 respectively). Family groups were difficult to identify because saline solution affected the quality of specimens.

Two families of isopods were identified, Armadillidiidae and Philosciidae, which are both terrestrial. Armadillidiidae was only found on A1 in very small numbers (2) and Philosciidae was only present on A1 (5 individuals) and A2 (19 individuals). One centipede (class Chilopoda) of the family Lithobiidae was found on A2, and an individual of the family Henicopidae was found on OM, but centipedes were absent from all other plots in August 2014.

Plot B2 was the richest site in terms of individual ground dwelling invertebrates (766) followed by plot B1 (680). Plot A1 had 628 individuals and A2 had 421. OM had the lowest number of ground-dwelling invertebrates (242 individuals). Although both Site B plots had the most individuals, family diversity was lower than on Site A plots.

Eight families were recorded on B1 and there were only six on B2. Twelve families were recorded on A1 and 10 were found on A2. Eight families were recorded on OM. Linyphiidae were proportionally high on the Site B plots (≥ 45% of invertebrates on each site), and Lycosidae were higher on the Site A plots, especially A2 (40%). Lycosidae were not evenly spread between quadrats on each plot, with numbers being highest in A1Q5 on A1 plots (Figure 5.2a) and A2Q3 on A2 plots (Figure 5.2b). Linyphiidae were in numbers > 40 in all quadrats on B1 (Figure 5.3a) and > 70 in all quadrats on B2, apart from in B2Q3 (Figure 5.3b). The distribution of Carabidae and Pulmonata were similar across Site A and Site B plots but were lower on OM (Figure 5.4).

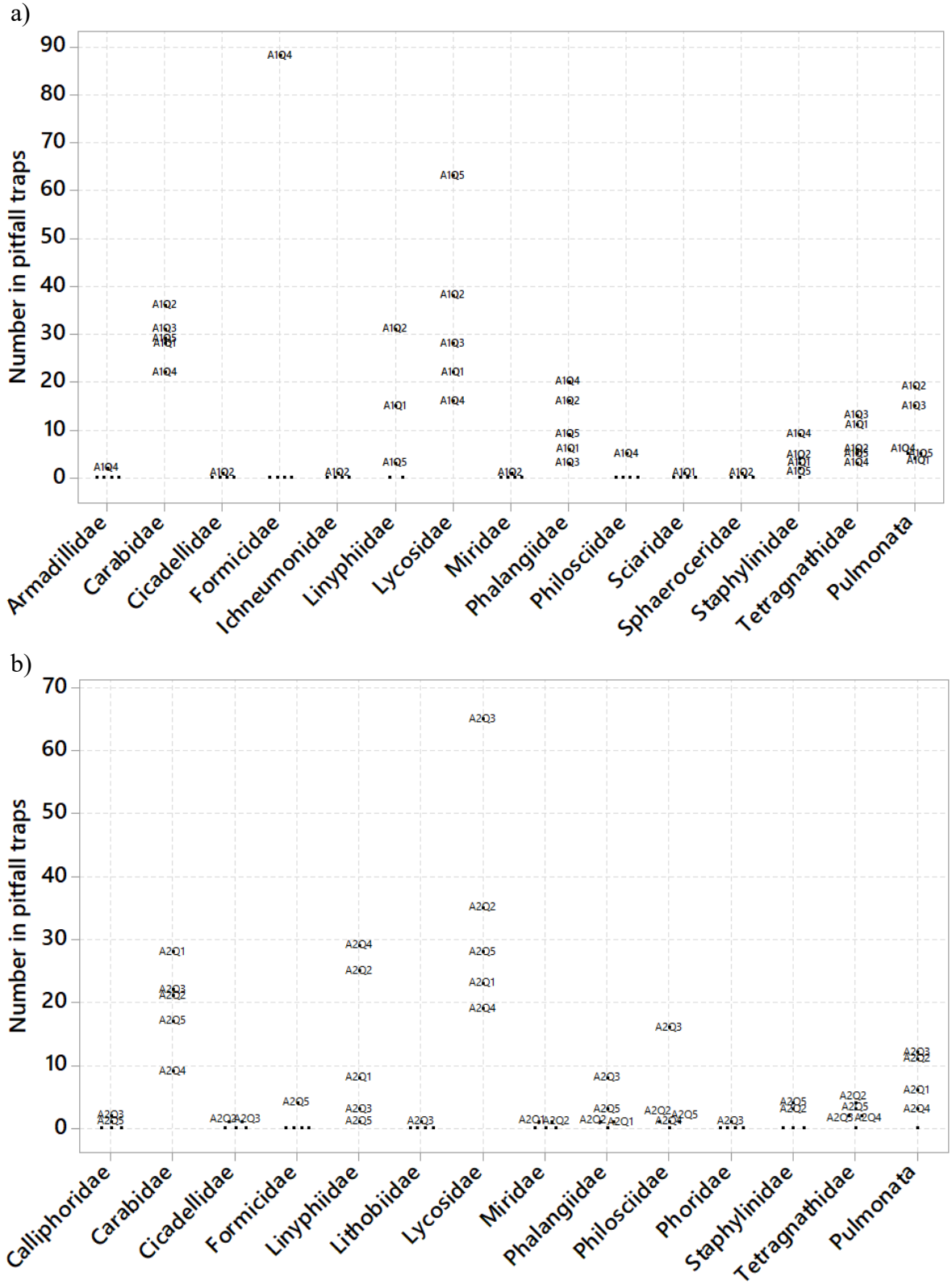


Figure 5.2 Number of invertebrates at family level caught in pitfall traps in August 2014 on a) A1 (Q1-Q5; n = 5) and b) A2 (Q1-Q5; n = 5).

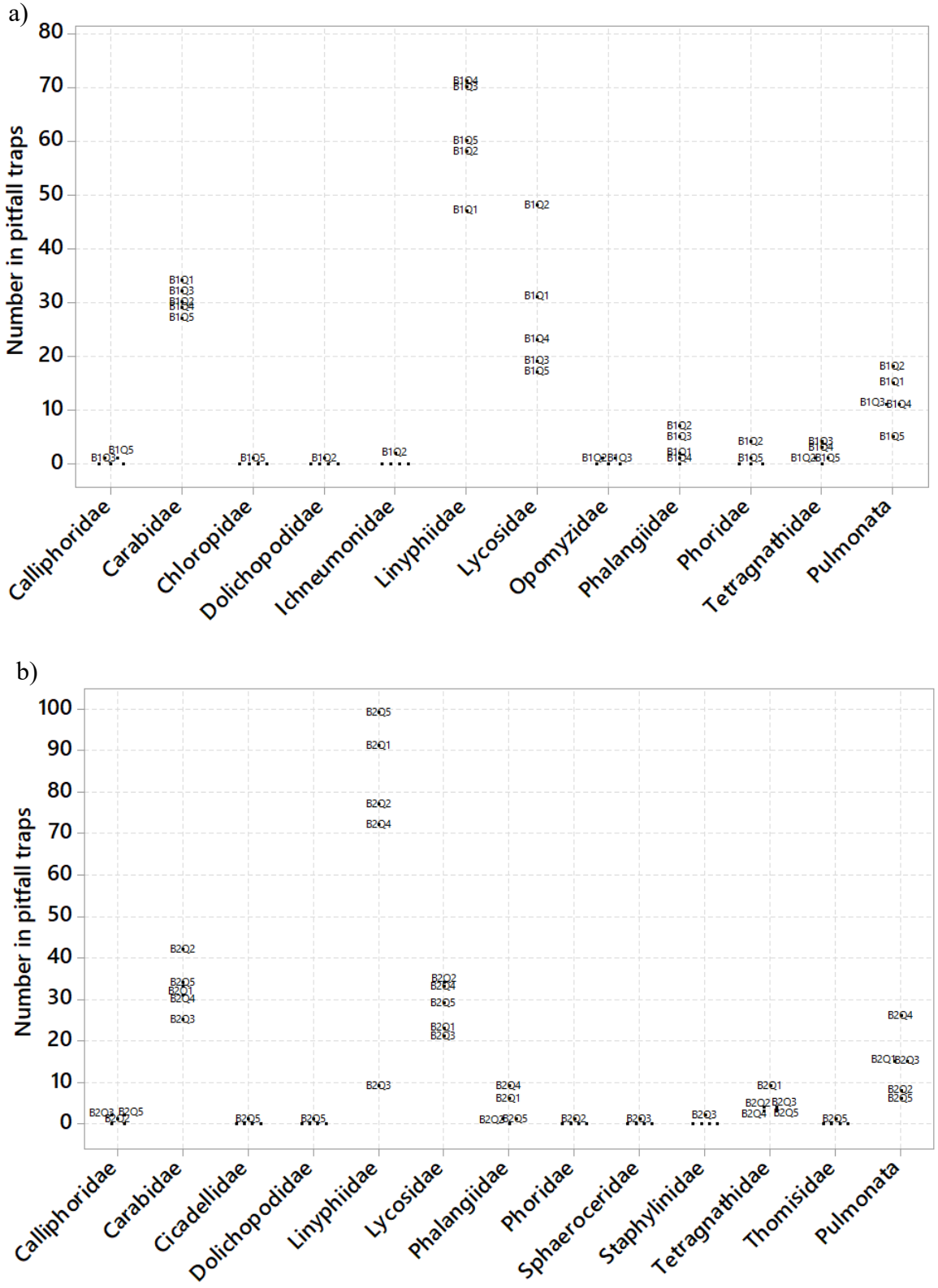


Figure 5.3 Number of invertebrates at family level caught in pitfall traps in August 2014 on a) B1 (Q1-Q5; n = 5) and b) B2 (Q1-Q5; n = 5).

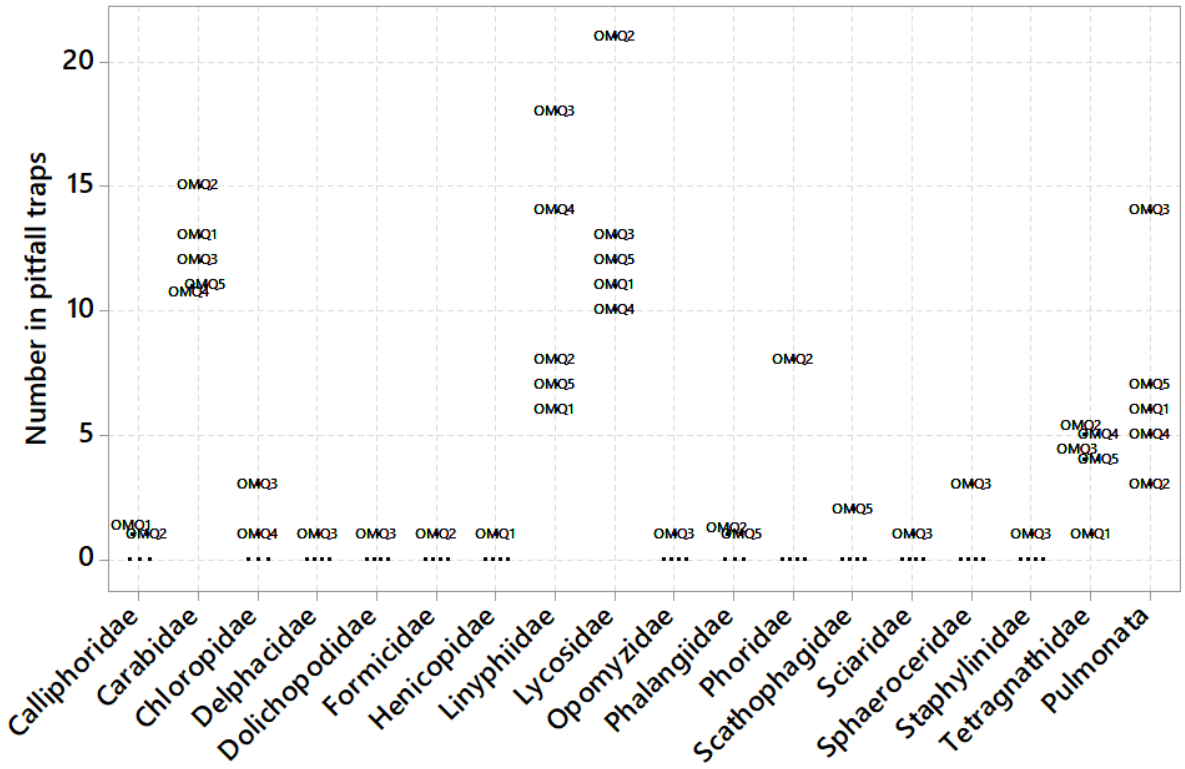


Figure 5.4 Number of invertebrates at family level caught in pitfall traps in August 2014 on OM (Q1-Q5; n = 5).

5.3.1.3 Steart Marsh and Otterhampton Marsh 2015 (post-breach)

In August 2015, 645 individuals were caught in pitfall traps across all plots on Steart Marsh and OM (Table 5.4). This was over 4000 individuals less than were caught on the SSSI plot in August 2016. Most individuals were caught on A1, A2 and OM. Numbers were substantially lower on B1 and B2, and family groups were also lower on these plots.

Table 5.4 Total invertebrate quantities and family groups caught in pitfall traps over 9 days on each study plot in August 2015. SSSI (collected in August 2016) is included for comparison. * includes the order Pulmonata.

	A1	A2	B1	B2	OM	SSSI
Total invertebrate quantity	182	218	29	48	168	c.4712
Family group quantity	14	13	6	8	17*	16

Linyphiidae and Lycosidae were the most prevalent members of the order Araneae caught in pitfall traps on A1 and A2, but there were only 25 individuals of Lycosidae in A1 and 41 in A2. This differs considerably from the number of this family caught pre-beach.

However, the most notable change was on the Site B plots. Pre-breach, Linyphiidae numbers were > 300 individuals on both arable plots, but one year after the breach there were only 4 individuals on B1 in quadrats B1Q1 and B1Q2 (Table 5.5; Figure 5.6a) and 2 individuals on B2 in quadrat B2Q4 (Table 5.5; Figure 5.6b). Pulmonate slugs persisted on OM but were no longer found on the other plots.

Carabid numbers were much lower one-year post-breach and were also absent from B1. Diptera numbers were higher in traps post-breach, especially specimens in the families Scathophagidae, and Sphaeroceridae, the latter being present on all plots. Members of the Tipulidae family were found on both Site A plots and OM, with the highest amount found on A2 (44 individuals). Most of the specimens were caught in quadrat A2Q4 (figure 5.5b). A higher number of Ichneumonid wasps were on the Site A plots post-breach, but only one member of the Hemiptera order was caught in traps in August 2015. This individual was from the family Cicadellidae and was caught on OM in quadrat OMQ3 (Figure 5.7).

Amphipods from the family Talitridae were caught on both Site A plots, although numbers were higher on A1 and in four of the five quadrats (Figure 5.5a) whereas they were only found in one quadrat on A2 (Figure 5.5b). Sphaeromatidae isopods were found in all plots, although numbers were highest on A1 and A2. These littoral families were absent pre-breach.

Table 5.5 Invertebrate families found in pitfall traps on Steart Marsh study plots (A1-B2) and Otterhampton Marsh in August 2015.

Class	Order	Family	A1 QTY	A2 QTY	B1 QTY	B2 QTY	OM QTY
Arachnida	Araneae	Linyphiidae	35	27	4	2	22
		Lycosidae	25	41	0	4	67
		Tetragnathidae	1	0	0	0	0
		Thomisidae	0	0	0	0	4
Gastropoda	Pulmonata	Unidentified slugs	0	0	0	0	21
Insecta	Coleoptera	Carabidae	5	2	0	1	17
		Hydrophilidae	0	0	0	0	2
		Staphylinidae	1	0	0	0	2
		Coccinellidae	0	0	0	0	1
	Diptera	Chloropidae	0	1	0	1	0
		Dolichopodidae	1	0	1	0	1
		Ephydridae	5	4	3	23	8
		Opomyzidae	0	0	0	0	1
		Phoridae	0	0	0	0	3
		Scathophagidae	15	8	3	5	0
		Sphaeroceridae	34	31	15	11	13
		Tipulidae	8	44	0	0	2
	Hemiptera	Cicadellidae	0	0	0	0	1
	Hymenoptera	Braconidae	0	1	0	0	0
		Ichneumonidae	2	20	0	0	0
		Eulophidae	0	0	0	0	1
		Formicidae	1	2	0	0	0
Malacostraca	Amphipoda	Talitridae	28	4	0	0	0
	Isopoda	Sphaeromatidae	21	33	3	1	2

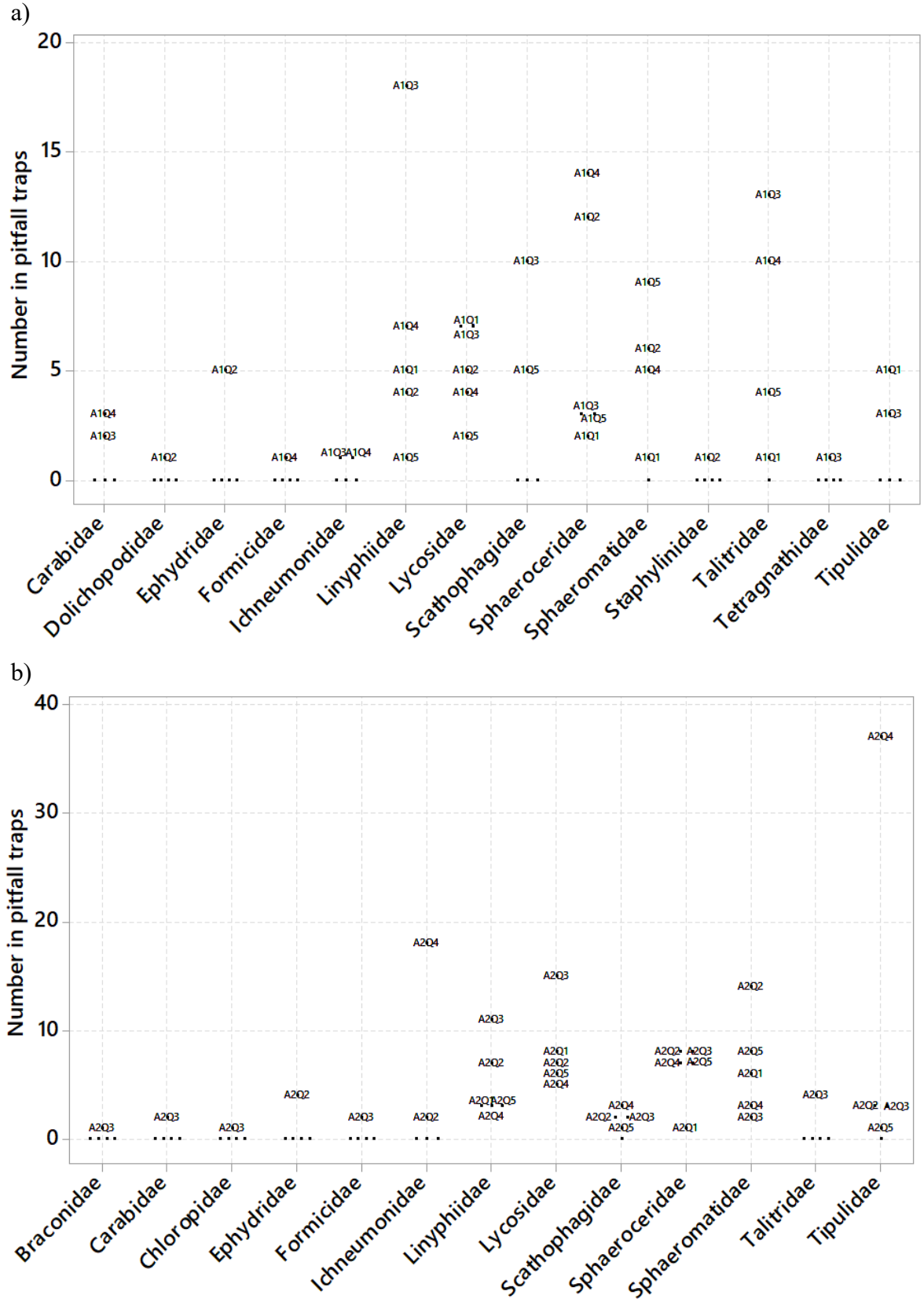


Figure 5.5 Number of invertebrates at family level caught in pitfall traps in August 2015 on a) A1 (Q1-Q5; n = 5) and b) A2 (Q1-Q5; n = 5). Quadrats that did not contain specimens of family groups are not labelled.

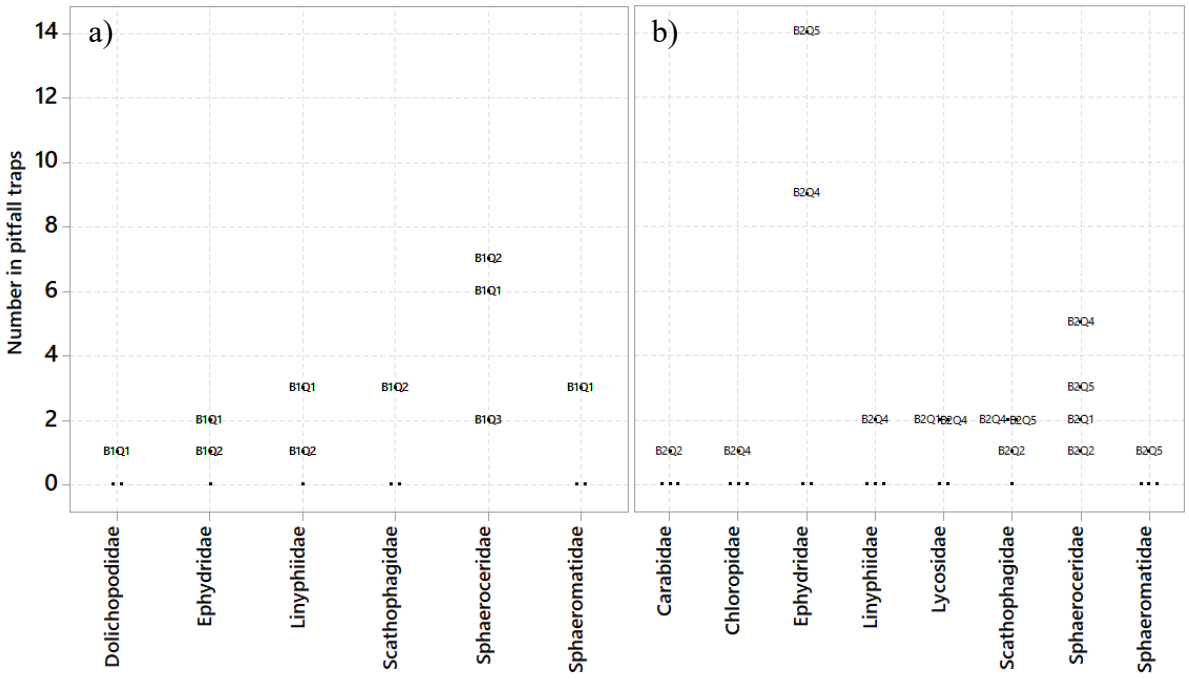


Figure 5.6 Number of invertebrates at family level caught in pitfall traps in August 2015 on a) B1 (Q1-Q3; n = 3) and b) B2 (Q1, Q2, Q4 and Q5; n = 4). Quadrats that had all pitfall traps flooded are not included because invertebrate specimens were not caught in these traps.

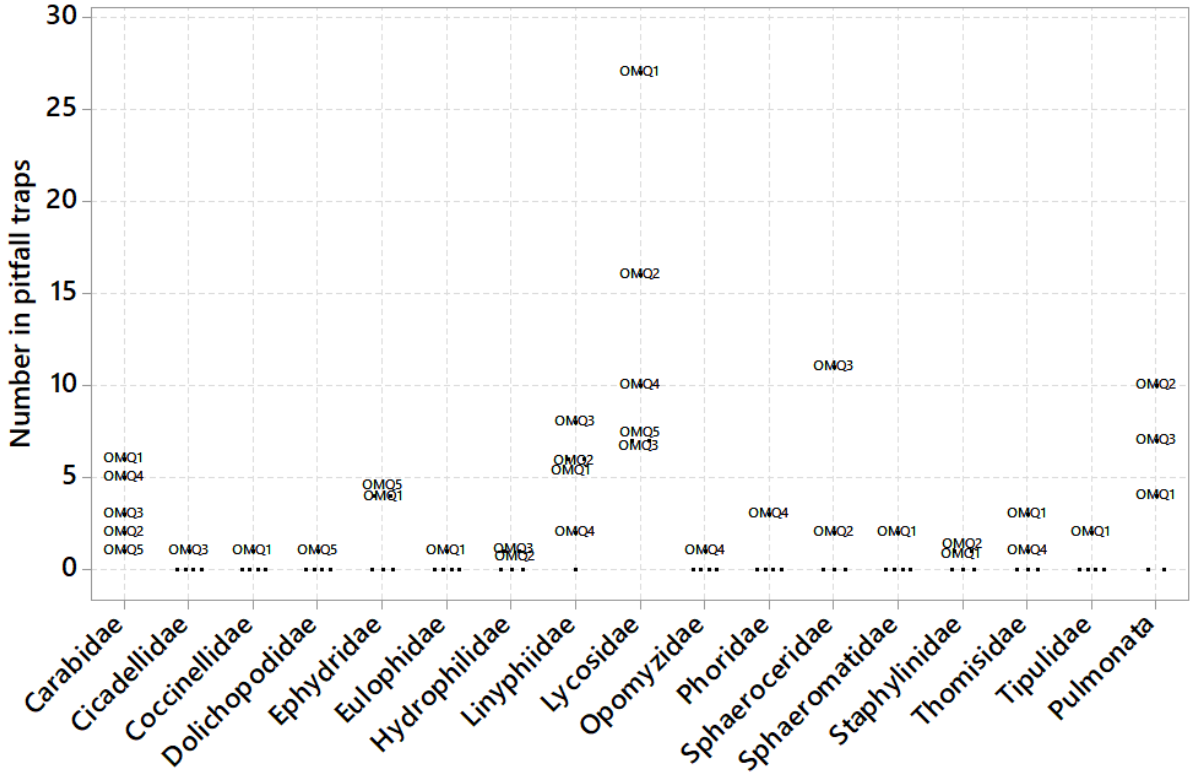


Figure 5.7 Number of invertebrates at family level caught in pitfall traps in August 2015 on OM (Q1-Q5; n = 5).

5.3.1.4 Steart Marsh and Otterhampton Marsh 2016 (post-breach)

Two years after the breach a smaller number (1,038 of individual invertebrates) were caught in pitfall traps (excluding the SSSI) from 28 families compared to pre-breach samples, but these numbers were greater than one year previously. Within the plots, post-breach, the specimen count was highest in A1 and A2 and there was a higher count in OM compared to both Site B plots (Table 5.6).

The most prevalent family found in pitfall traps on the Site A plots was Talitridae. The numbers on A1 and A2 had increased to 241 and 245 individuals respectively (Table 5.7; Figure 5.8). Numbers were still very low on B1 (5 individuals) and amphipods were absent from B2 and OM (Table 5.7; Figure 5.9 and Figure 5.10).

Table 5.6 Total invertebrate quantities and family groups caught in pitfall traps over 9 days on each study plot in August 2016. SSSI (collected in August 2016) is included for comparison.

	A1	A2	B1	B2	OM	SSSI
Total invertebrate quantity	398	460	48	43	89	c.4712
Family group quantity	15	21	11	7	13	16

In August 2016, a shore crab specimen (*Carcinus maenas*) from the family Portunidae was caught in a pitfall trap on B2 (Table 5.7; Figure 5.9b). This species was not found on the SSSI during this study. Numbers of Sphaeromatidae isopods caught within traps were lower in comparison to the previous year's sample.

Table 5.7 Invertebrate families caught in pitfall traps on Steart Marsh study plots (A1-B2) and Otterhampton Marsh in August 2016.

Class	Order	Family	A1 QTY	A2 QTY	B1 QTY	B2 QTY	OM QTY
Arachnida	Araneae	Linyphiidae	67	80	22	23	24
		Lycosidae	3	15	4	4	29
		Tetragnathidae	6	2	0	0	1
		Thomisidae	0	0	0	0	1
	Opiliones	Phalangiidae	0	0	0	0	3
Insecta	Coleoptera	Anthicidae	0	0	0	0	1
		Carabidae	11	19	0	1	14
		Coccinellidae	0	1	1	0	5
		Latridiidae	0	0	0	0	1
	Diptera	Bibionidae	0	0	0	0	3
		Chloropidae	4	3	1	0	0
		Dolichopodidae	0	1	0	0	0
		Ephydridae	0	1	0	0	0
		Scathophagidae	23	14	2	0	0
		Sciaridae	0	1	3	1	5
		Sphaeroceridae	7	43	2	3	0
		Stratiomyidae	0	0	0	0	1
		Tipulidae	2	6	4	10	0
	Hemiptera	Cicadellidae	0	1	0	0	0
		Miridae	4	1	0	0	0
		Saldidae	7	8	3	0	0
	Hymenoptera	Braconidae	2	2	0	0	0
		Ichneumonidae	1	9	1	0	0
		Figitidae	0	1	0	0	0
		Formicidae	1	1	0	0	1
Malacostraca	Amphipoda	Talitridae	241	245	5	0	0
	Decapoda	Portunidae	0	0	0	1	0
	Isopoda	Sphaeromatidae	7	6	0	0	0

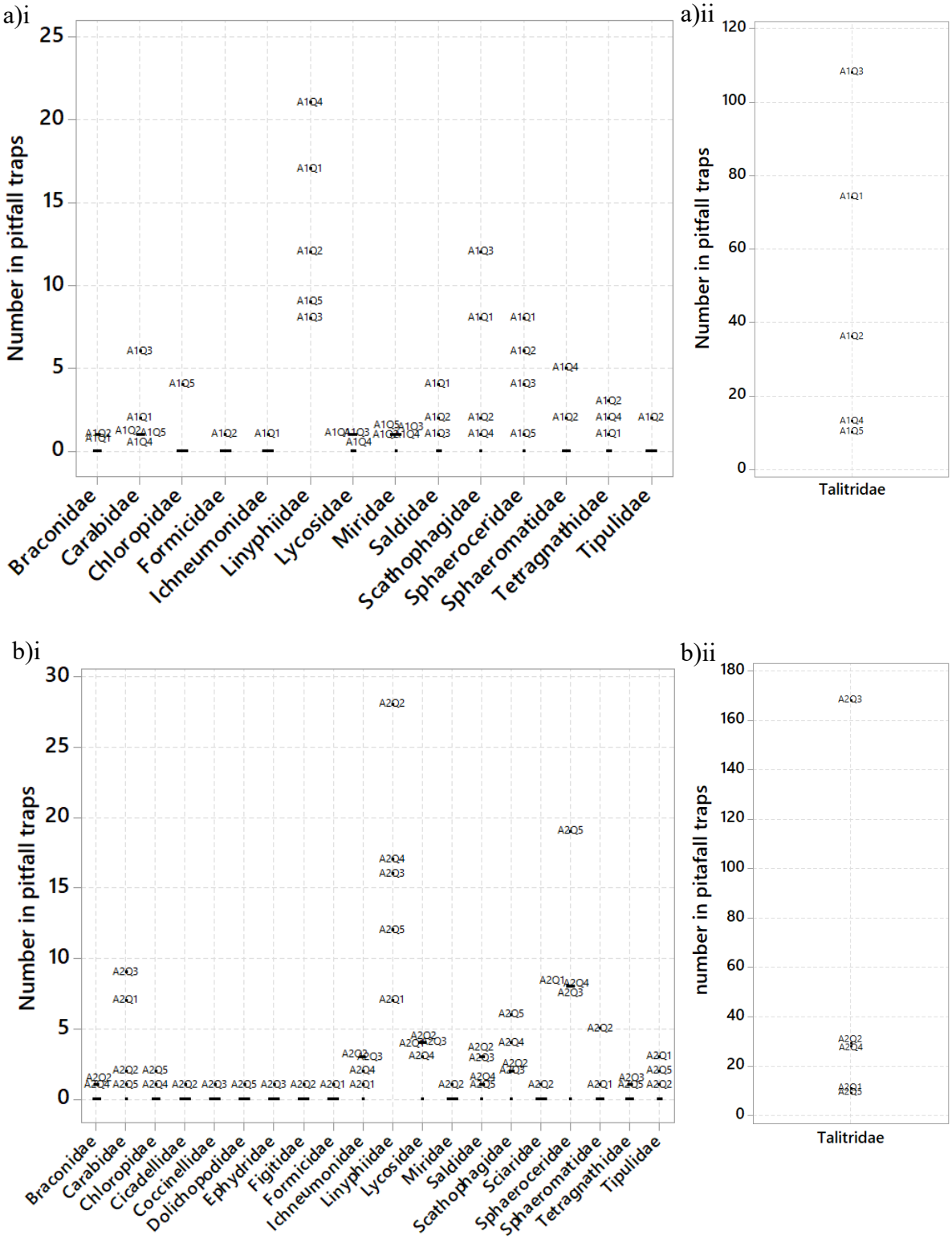


Figure 5.8 Number of invertebrates at family level caught in pitfall traps in August 2016 on a) A1 (Q1-Q5; n = 5) and b) A2 (Q1-Q5; n = 5); a)i is all invertebrates caught on A1 in August 2016 apart from Talitridae; a)ii is Talitridae caught in August 2016 on A1; b)i is all invertebrates caught on A2 in August 2016 apart from Talitridae; b)ii is Talitridae caught in August 2016 on A2.

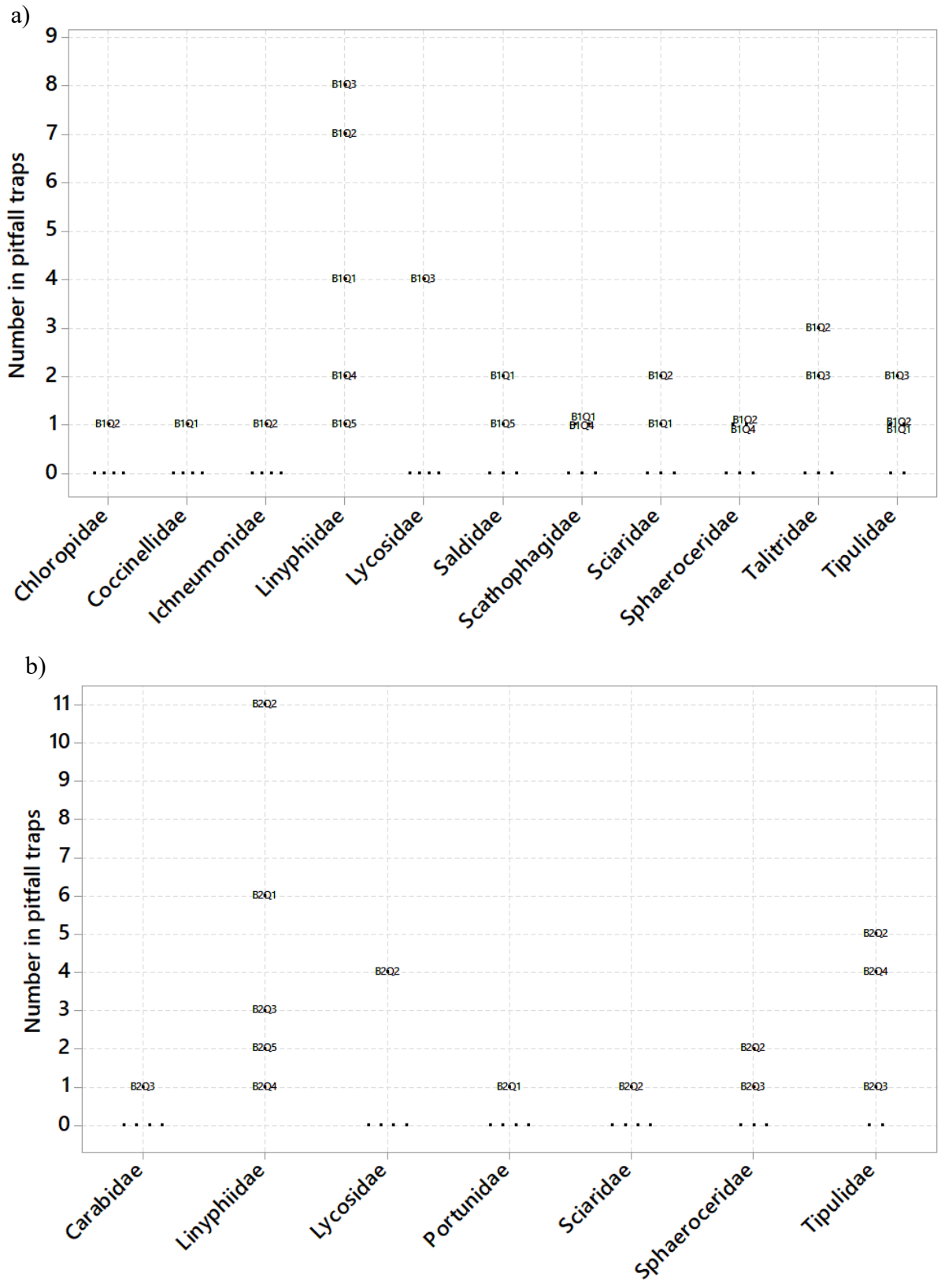


Figure 5.9 Number of invertebrates at family level caught in pitfall traps in August 2016 on a) B1 (Q1-Q5; n = 5) and b) B2 (Q1-Q5; n = 5).

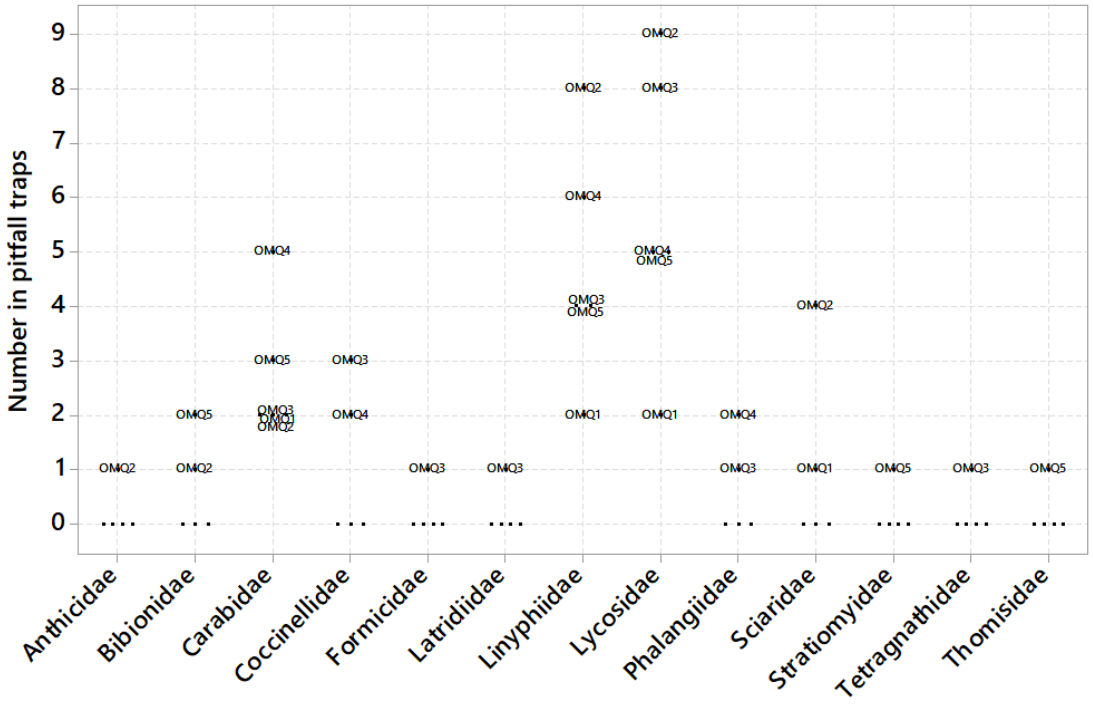


Figure 5.10 Number of invertebrates at family level caught in pitfall traps in August 2016 on OM (Q1-Q5; n = 5).

5.3.1.5 Steart Marsh and Otterhampton Marsh 2017 (post-breach)

Three years after the breach invertebrate specimen numbers caught in pitfall traps had increased across all sites, to ≥680 on the Site A plots, with smaller counts on the Site B plots and OM (Table 5.8).

Table 5.8 Total invertebrate quantities and family groups caught in pitfall traps over 9 days on each study plot in August 2017. SSSI (collected in August 2016) is included for comparison.

	A1	A2	B1	B2	OM	SSSI
Total invertebrate quantity	693	680	139	117	245	c.4712
Family group quantity	19	20	13	10	20	16

More than 450 Amphipods were caught on both Site A plots and 67 individuals were caught on B1. Numbers were still low on B2 and they were absent from OM. The highest numbers were caught in A1Q5 on A1 (> 200 individuals) (Figure 5.11a) and in A2Q4 on A2 (> 200 individuals) (Figure 5.11b). Scathophagidae flies were also relatively abundant in A2Q4. The highest number of Talitridae on the Site B plots was in B2Q1, but they were not caught in one of the quadrats on that plot (Figure 5.12b). By 2017 non-biting midge (Chironomidae) numbers were very low on all sites and more comparable to the SSSI. However, biting midge (Ceratopogonidae) numbers had increased further, especially on the Site A plots. More than 100 Stratiomyidae larvae were caught on OM, mainly in OMQ4 and OMQ5 (Figure 5.13),

but this family was absent from all other plots (Table 5.9). Saldidae (shore bug) numbers had also started to steadily increase from the previous year on all plots, and were found in all quadrats on A2, B1 and B2 (Figure 5.11b and Figure 5.12), but specimens from this family were not found on the SSSI.

Table 5.9 Invertebrate families caught in pitfall traps on Steart Marsh study plots (A1-B2) and Otterhampton Marsh in August 2017.

Class	Order	Family	A1 QTY	A2 QTY	B1 QTY	B2 QTY	OM QTY
Arachnida	Araneae	Clubionidae	1	0	0	0	0
		Linyphiidae	12	14	10	36	20
		Lycosidae	17	6	8	3	15
		Tetragnathidae	1	4	0	0	1
		Thomisidae	0	0	0	0	1
Gastropoda	Ellobiida	Ellobiidae	11	11	0	0	0
	Littorinimorpha	Hydrobiidae	1	3	0	0	0
Insecta	Coleoptera	Carabidae	4	5	1	3	1
	Diptera	Bibionidae	0	5	0	0	0
		Calliphoridae	0	0	0	0	1
		Chironomidae	0	0	0	0	2
		Chloropidae	0	3	2	6	0
		Dolichopodidae	0	1	0	0	1
		Drosophilidae	0	1	0	0	0
		Ephydridae	0	0	0	0	50
		Scathophagidae	37	76	3	3	26
		Sciaridae	0	0	0	0	3
		Sphaeroceridae	31	39	10	11	4
		Stratiomyidae	0	0	0	0	107
		Tipulidae	13	1	1	0	1
	Hemiptera	Aphididae	1	0	0	0	0
		Cicadellidae	32	4	6	2	0
		Delphacidae	15	22	11	6	1
		Miridae	0	0	4	0	0
		Psyllidae	0	0	0	0	1
		Saldidae	9	13	15	35	5
	Hymenoptera	Apidae	0	0	0	0	1
		Braconidae	0	0	0	0	1
		Ichneumonidae	4	4	0	0	0
		Formicidae	0	1	0	0	0
		Pteromalidae	1	0	0	0	1
Malacostraca	Amphipoda	Talitridae	498	460	67	12	0
	Decapoda	Portunidae	1	0	0	0	0
	Isopoda	Sphaeromatidae	4	7	1	0	0

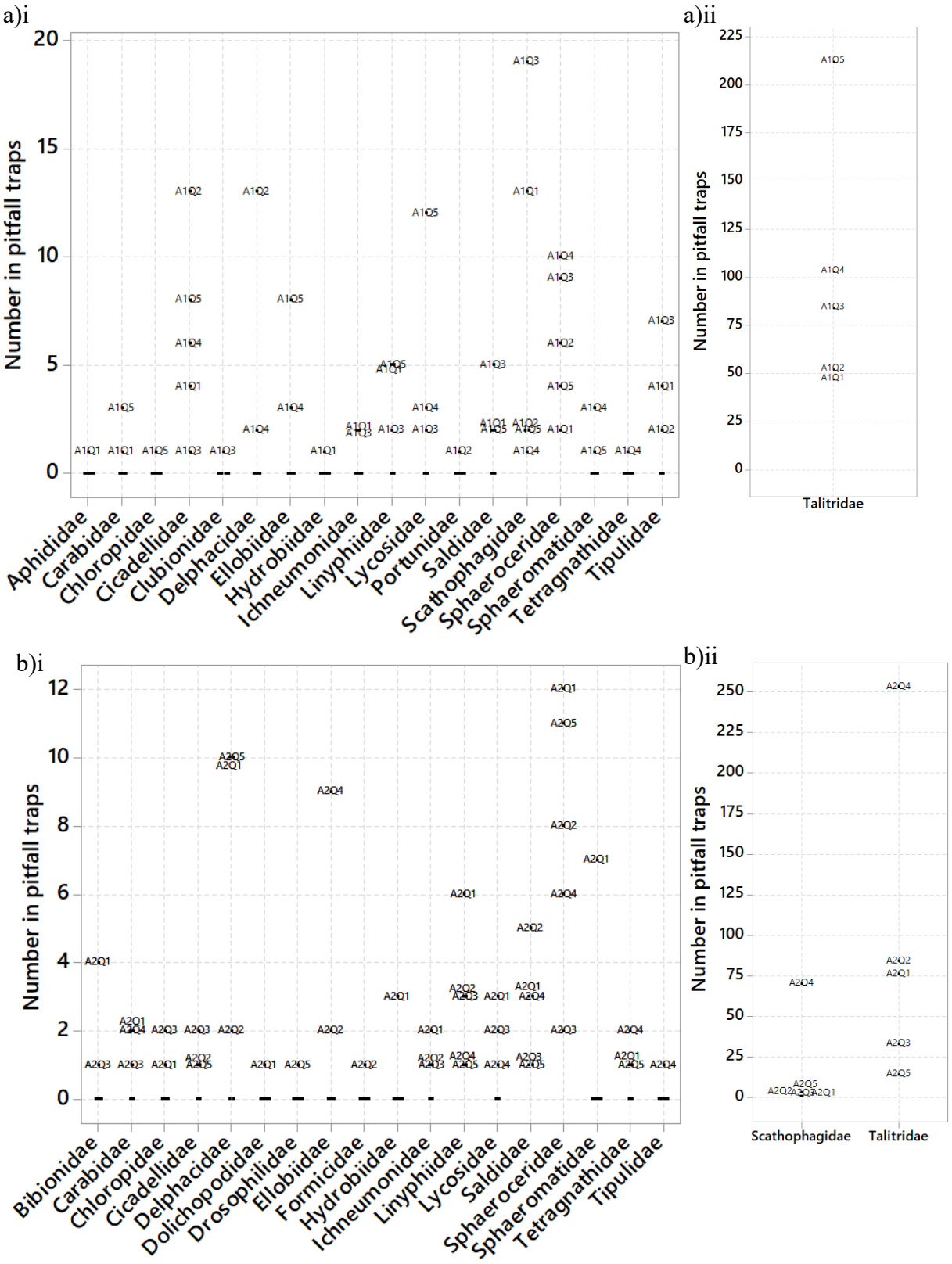


Figure 5.11 Number of invertebrates at family level caught in pitfall traps in August 2017 on a)i A1 (Q1-Q5; n = 5) and b)i A2 (Q1-Q5; n = 5). a)ii is Talitridae caught in August 2017 on A1; b)ii is Scathophagidae and Talitridae caught in August 2017 on A2.

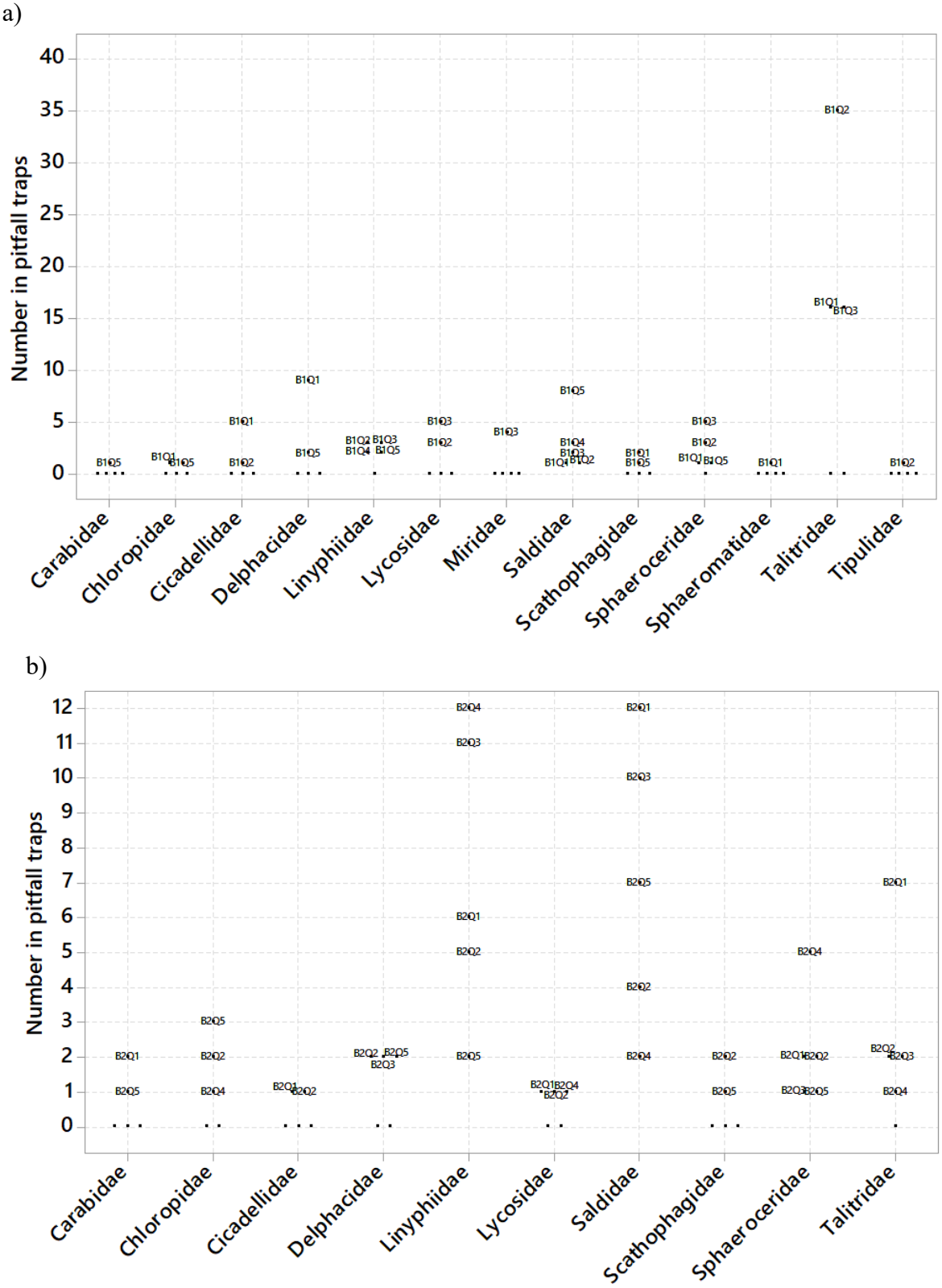


Figure 5.12 Number of invertebrates at family level caught in pitfall traps in August 2017 on a) B1 (Q1-Q5; n = 5) and b) B2 (Q1-Q5; n = 5).

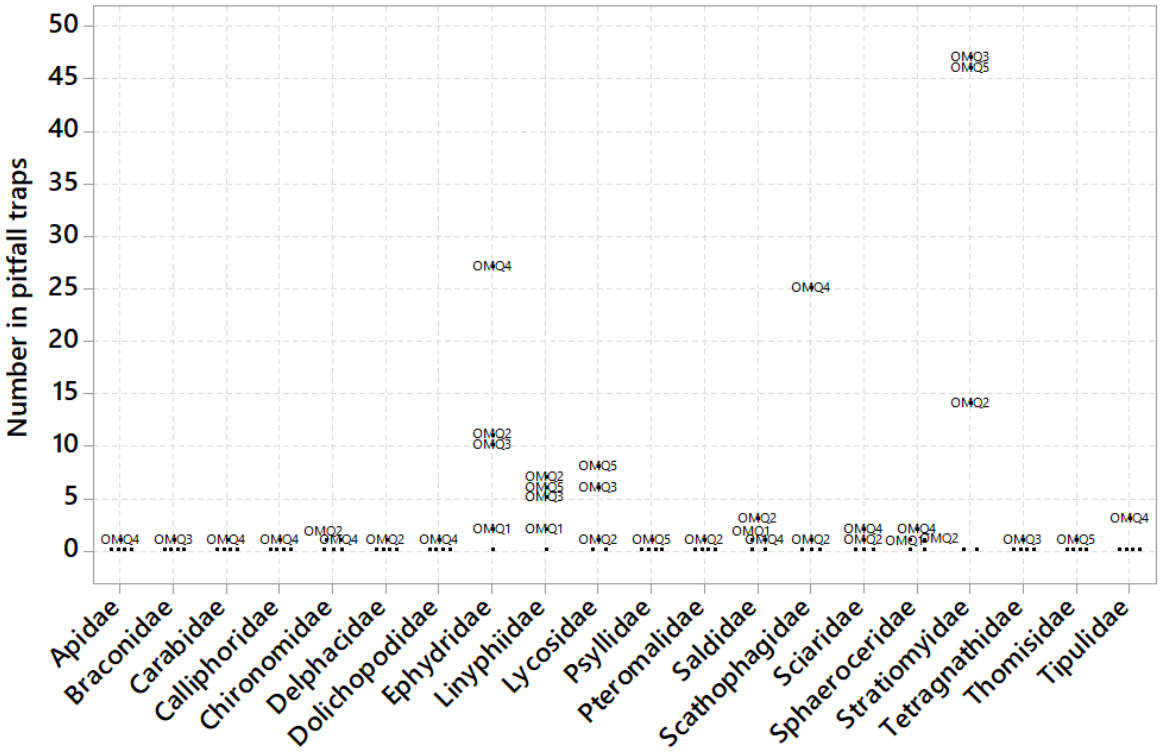


Figure 5.13 Number of invertebrates at family level caught in pitfall traps in August 2017 on OM (Q1-Q5; n = 5).

5.3.2 Plant-dwelling invertebrates

5.3.2.1 SSSI target state

The most abundant plant-feeding invertebrate taxa caught using sweep-netting on the SSSI were in the families Delphacidae (Hemiptera) and Tephritidae (Diptera) (Table 5.10).

Table 5.10 Invertebrate families found in sweep nets on the SSSI study plot in August 2016.

Class	Order	Family (Subfamily)	SSSI QTY
Insecta	Diptera	Ceratopogonidae	1
		Chironomidae	1
		Chloropidae	3
		Lonchaeidae	1
		Phoridae	3
		Sciomyzidae	1
		Sepsidae	4
		Sphaeroceridae	1
		Tephritidae	98
		Tipulidae	2
	Hemiptera	Aphididae	1
		Cicadellidae	1
		Cixiidae	8
		Delphacidae	106
		Miridae	1
	Hymenoptera	Braconidae	1
		Eulophidae	1
		Figitidae	1
		Pteromalidae	4
	Trichoptera	Limnephilidae	3

Delphacidae were most numerous in SSSIQ2, but specimens were found in all quadrats on the SSSI. Tephritidae were also found in each quadrat, but they were most numerous in SSSIQ4. All other specimens within family groups were low in numbers, and absent from some quadrats compared to Delpahcidae and Tephritidae (Figure 5.14).

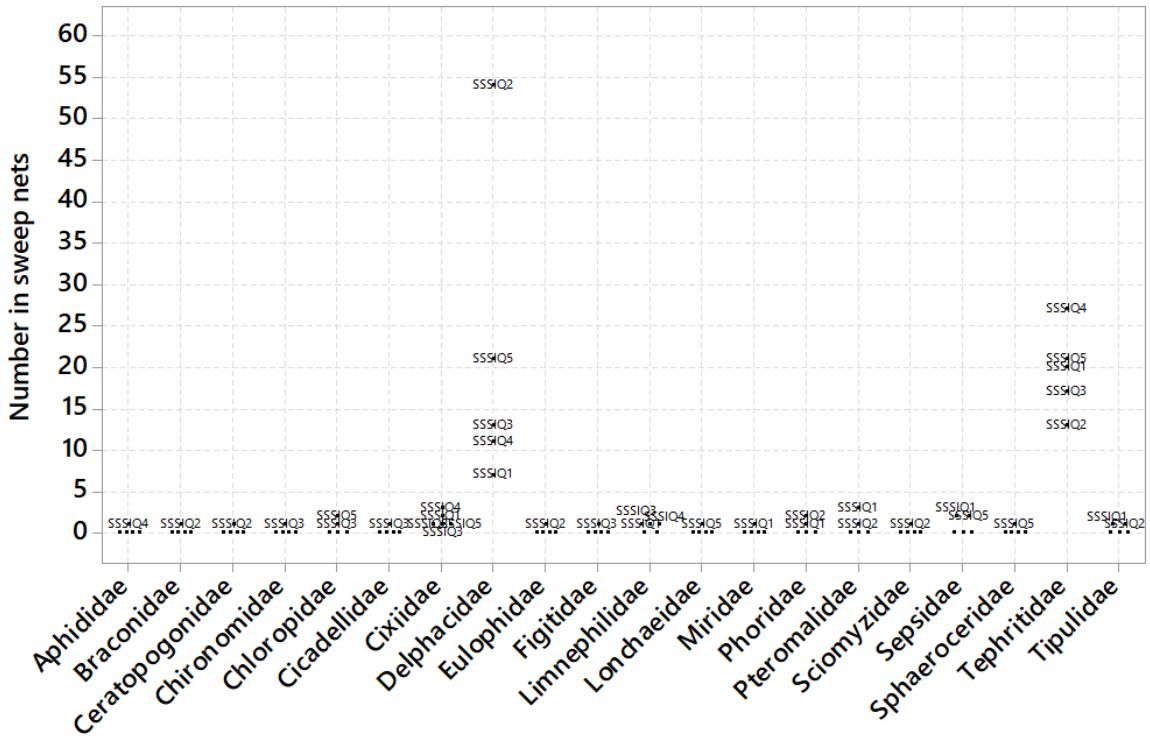


Figure 5.14 Number of invertebrates at family level caught in sweep nets on the SSSI saltmarsh (Q1-Q5; n = 5).

5.3.2.2 Steart Marsh and Otterhampton Marsh August 2014 (pre-breach)

In August 2014, 238 individual invertebrates were caught in sweep nets across all study plots (A1-OM; excluding the SSSI). These numbers were not evenly spread across each study plot, with the highest numbers being caught in plot A2 and the lowest in OM (Table 5.11).

Table 5.11 Total invertebrate quantities and family groups caught in sweep nets on each study plot in August 2014. SSSI (collected in August 2016) is included for comparison.

	A1	A2	B1	B2	OM	SSSI
Total invertebrate quantity	46	90	41	39	22	242
Family group quantity	10	23	14	15	11	20

Families representative of the orders Araneae, Hemiptera and Diptera were the most abundant in the sweep nets pre-breach (Table 5.12).

Table 5.12 Invertebrate families caught in sweep-nets on Steart Marsh study plots (A1-B2) and Otterhampton Marsh in August 2014.

Class	Order	Family	A1 QTY	A2 QTY	B1 QTY	B2 QTY	OM QTY
Arachnida	Araneae	Araneidae	3	5	3	13	1
		Linyphiidae	0	0	1	2	1
		Lycosidae	0	1	0	0	0
		Tetragnathidae	3	7	2	1	0
Insecta	Coleoptera	Coccinellidae	0	1	0	0	0
		Latridiidae	0	0	0	1	1
		Staphylinidae	0	0	0	0	1
	Diptera	Anthomyzidae	0	0	1	0	0
		Bibionidae	0	2	0	0	0
		Cecidomyiidae	0	0	3	0	0
		Ceratopogonidae	0	2	0	0	0
		Chironomidae	12	15	11	6	3
		Chloropidae	8	19	2	1	5
		Dolichopodidae	0	0	0	1	0
		Dryomyzidae	0	0	1	0	0
		Ephydridae	0	0	0	0	4
		Lonchopteridae	0	9	0	3	0
		Opomyzidae	0	1	0	2	0
		Phoridae	0	1	1	1	0
		Pipunculidae	0	1	0	0	0
		Scathophagidae	0	1	0	1	2
		Sciaridae	0	1	5	1	1
		Sepsidae	0	1	0	0	0
		Syrphidae	0	1	0	0	0
	Hemiptera	Aphrophoridae	13	13	7	1	0
		Cicadellidae	0	1	0	0	1
		Delphacidae	0	3	1	1	2
		Miridae	0	1	0	0	0
	Hymenoptera	Crabronidae	1	0	0	0	0
		Eulophidae	2	0	0	0	0
		Ichneumonidae	2	1	2	0	0
		Platygastridae	0	0	0	1	0
		Proctotrupidae	1	1	0	0	0
	Odonata	Coenagrionidae	0	0	1	0	0
	Orthoptera	Acrididae	1	2	0	0	0

Orb-weaver spiders (Araneidae) were present on all plots, with the highest number recorded in B2 (13 individuals) and the lowest in OM (1). An individual of the Lycosidae was found on A2, but the lack of presence of members of this family on other plots is not unexpected because they are typically ground-dwelling species. Linyphidae were not found on A1 and A2, but they were present in small numbers on B1 (1), B2 (2) and OM (1). Tetragnathidae were present on all plots apart from OM, with the highest number being found on A2 (7).

The most common Hempiteran family found in the sweep nets was Aphrophoridae (froghoppers), which was found on all plots apart from OM. Thirteen individuals of the family were found on both A1 and A2, but lower numbers were recorded on B1 (7) and B2 (1). Few grasshoppers (order Orthoptera) were found in the sweep nets, but members of the family Acrididae were present on A1 (1) and A2 (2). Representatives of Coleoptera were very low in sweep nets, with one individual of the family Coccinellidae found on A2 and a single individual of the family Staphylinidae found on OM. Latridiidae was found on B2 and OM.

Members of the Hymenoptera were caught in small numbers on A1 and A2 (Figure 5.15) and B1 but were absent on B2 (Figure 5.16) and OM (Figure 5.17). Ichneumonids were found in marginally higher numbers than other families, but numbers were relatively low compared to those found in Aranaea, Diptera and Hemiptera orders.

Of the Diptera, the most common families were Chironomidae (non-biting midges), and Chloropidae, especially on Site A plots. Chironomidae were found in each quadrat on A1 (Figure 5.15a) but were not trapped in A2Q5 (Figure 5.15b).

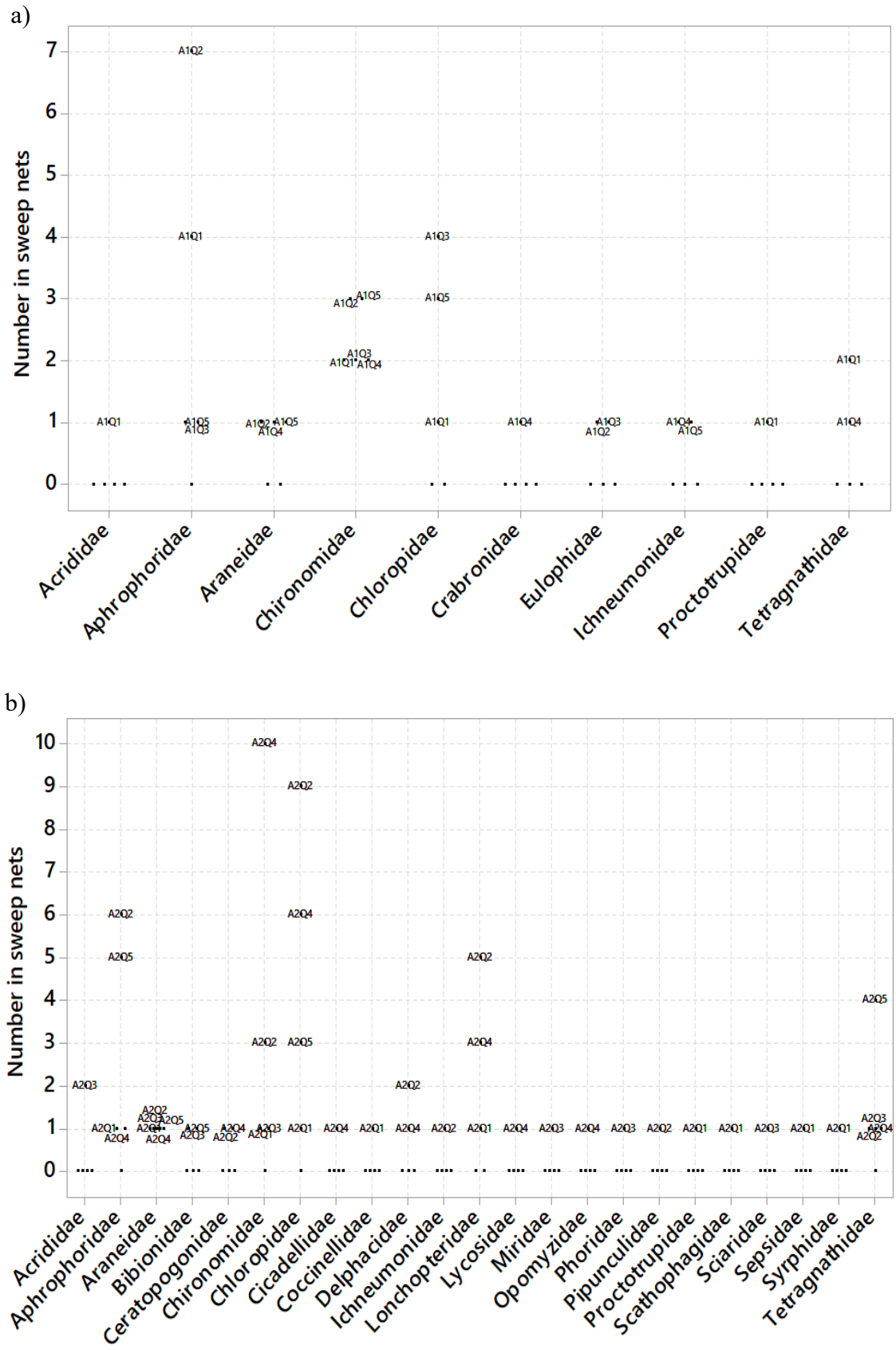


Figure 5.15 Number of invertebrates at family level caught in sweep nets in August 2014 on a) A1 (Q1-Q5; n = 5) and b) A2 (Q1-Q5; n = 5).

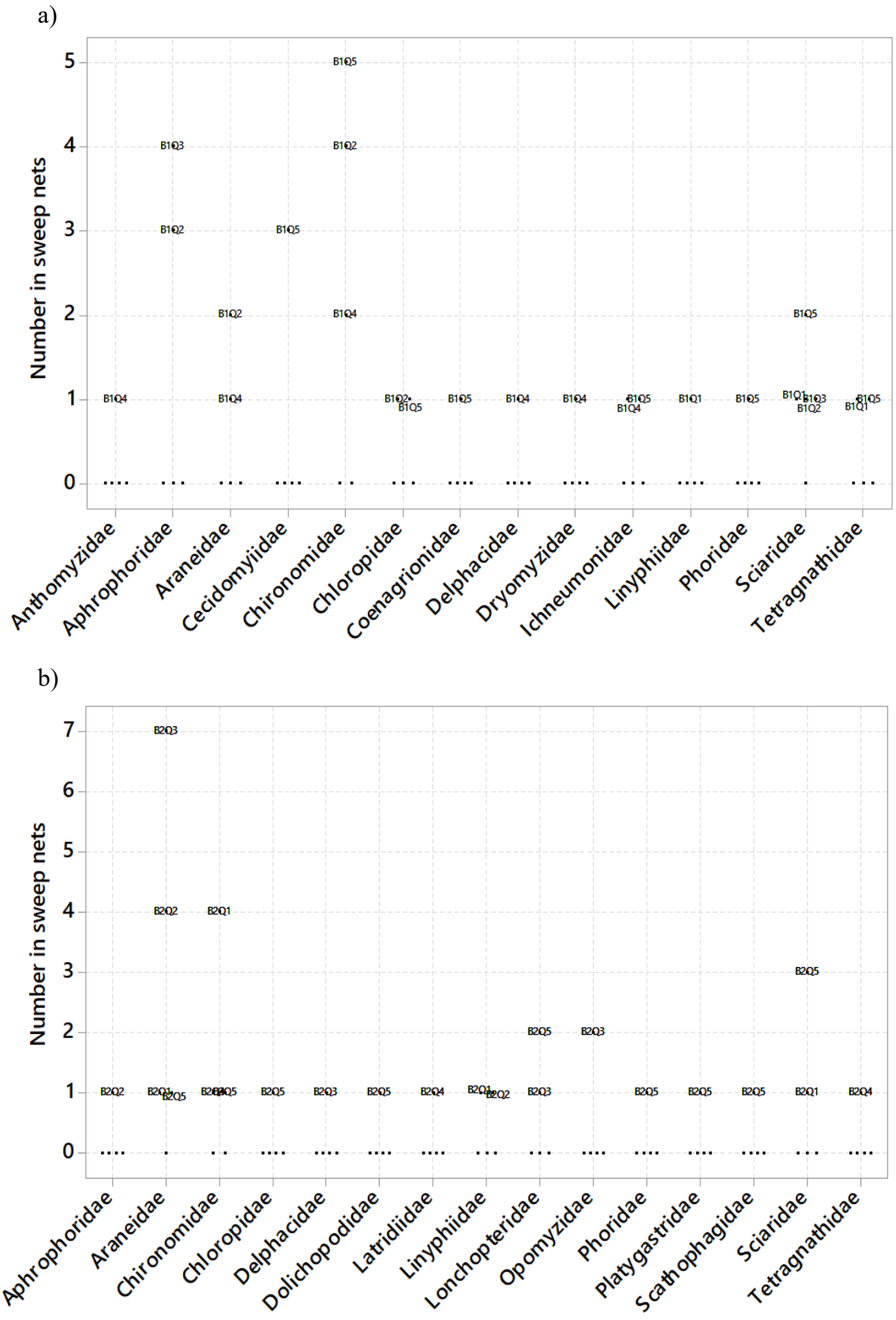


Figure 5.16 Number of invertebrates at family level caught in sweep nets in August 2014 on a) B1 (Q1-Q5; n = 5) and b) B2 (Q1-Q5; n = 5).

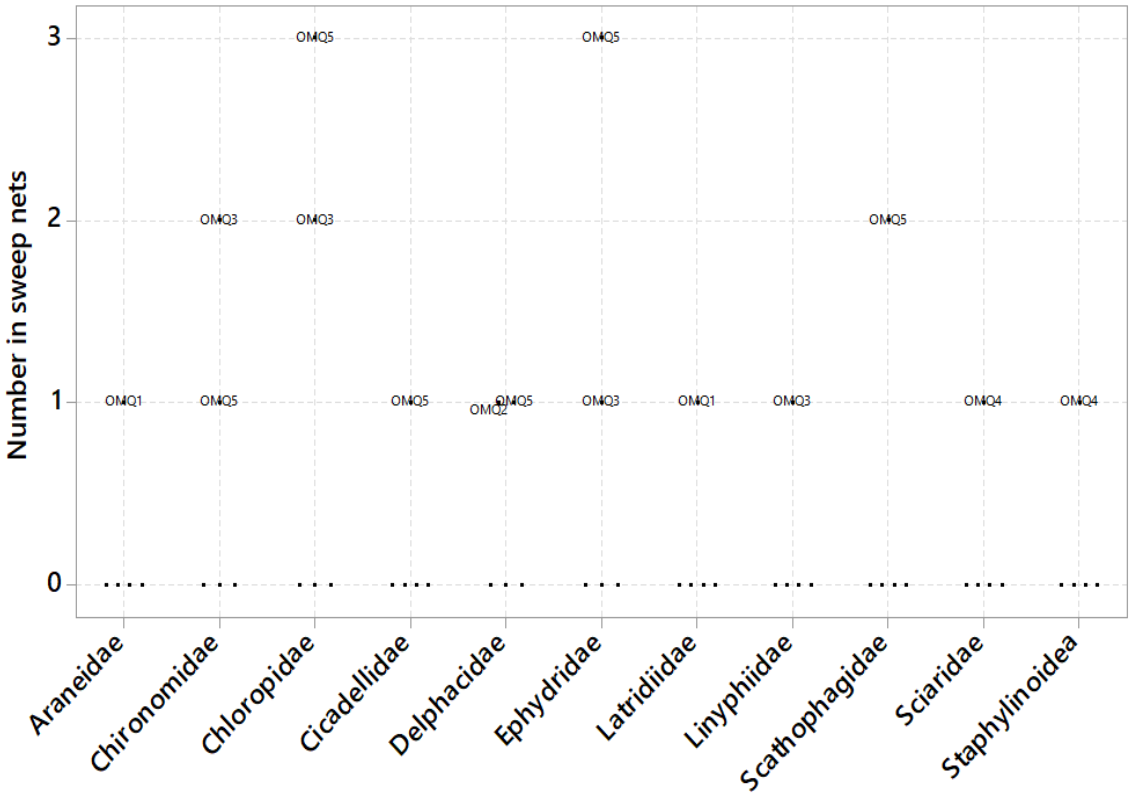


Figure 5.17 Number of invertebrates at family level caught in sweep nets in August 2014 on OM (Q1-Q5; n = 5).

5.3.2.3 Steart Marsh and Otterhampton Marsh August 2015 (post-breach)

One year after the breach, 373 individual specimens were caught in sweep nets on Steart Marsh and OM (Table 5.13).

Table 5.13 Total invertebrate quantities and family groups caught in sweep nets on each study plot in August 2015. SSSI (collected in August 2016) is included for comparison.

	A1	A2	B1	B2	OM	SSSI
Total invertebrate quantity	92	58	54	30	139	242
Family group quantity	19	16	5	5	11	20

Most specimens were from the Diptera families, Sphaeroceridae and Ephydridae (shore files), although Chironomidae were present on OM in higher numbers compared to the other study sites (Table 5.14; Figures 5.18-5.20). The number of family groups was highest on the Site A plots. Araneae numbers were low in sweep-nets in 2015.

Table 5.14 Invertebrate families found in sweep-nets on Steart Marsh study plots (A1-B2) and Otterhampton Marsh in August 2015.

Class	Order	Family (Subfamily)	A1 QTY	A2 QTY	B1 QTY	B2 QTY	OM QTY
Arachnida	Araneae	Araneidae	8	1	0	3	3
		Linyphiidae	5	2	0	0	1
		Lycosidae	1	0	0	0	0
		Tetragnathidae	0	0	0	0	5
		Thomisidae	0	0	0	0	5
Insecta	Coleoptera	Chrysomelidae	1	1	0	0	0
	Diptera	Anisopodidae	2	0	0	0	0
		Ceratopogonidae	0	3	0	0	0
		Chironomidae	3	2	1	0	56
		Chloropidae	2	0	1	0	0
		Culicidae	0	1	0	0	0
		Dolichopodidae	0	1	0	0	0
		Drosophilidae	2	1	0	0	0
		Ephydridae	12	4	28	13	39
		Opomyzidae	0	0	0	0	1
		Scathophagidae	0	2	0	0	0
		Simuliidae	1	0	0	0	0
		Sphaeroceridae	40	35	23	12	23
		Syrphidae	0	1	0	0	0
		Tipulidae	7	0	0	1	4
	Hemiptera	Delphacidae	1	0	0	0	0
		Miridae	1	0	0	0	0
	Hymenoptera	Braconidae	2	1	0	0	0
		Eulophidae	0	0	0	1	0
		Eurytomidae	0	1	0	0	0
		Figitidae	0	0	0	0	1
		Ichneumonidae	1	1	0	0	0
		Megaspilidae	1	0	0	0	0
		Mymaridae	0	0	0	0	1
		Pteromalidae	1	0	1	0	0
	Psocoptera		1	1	0	0	0

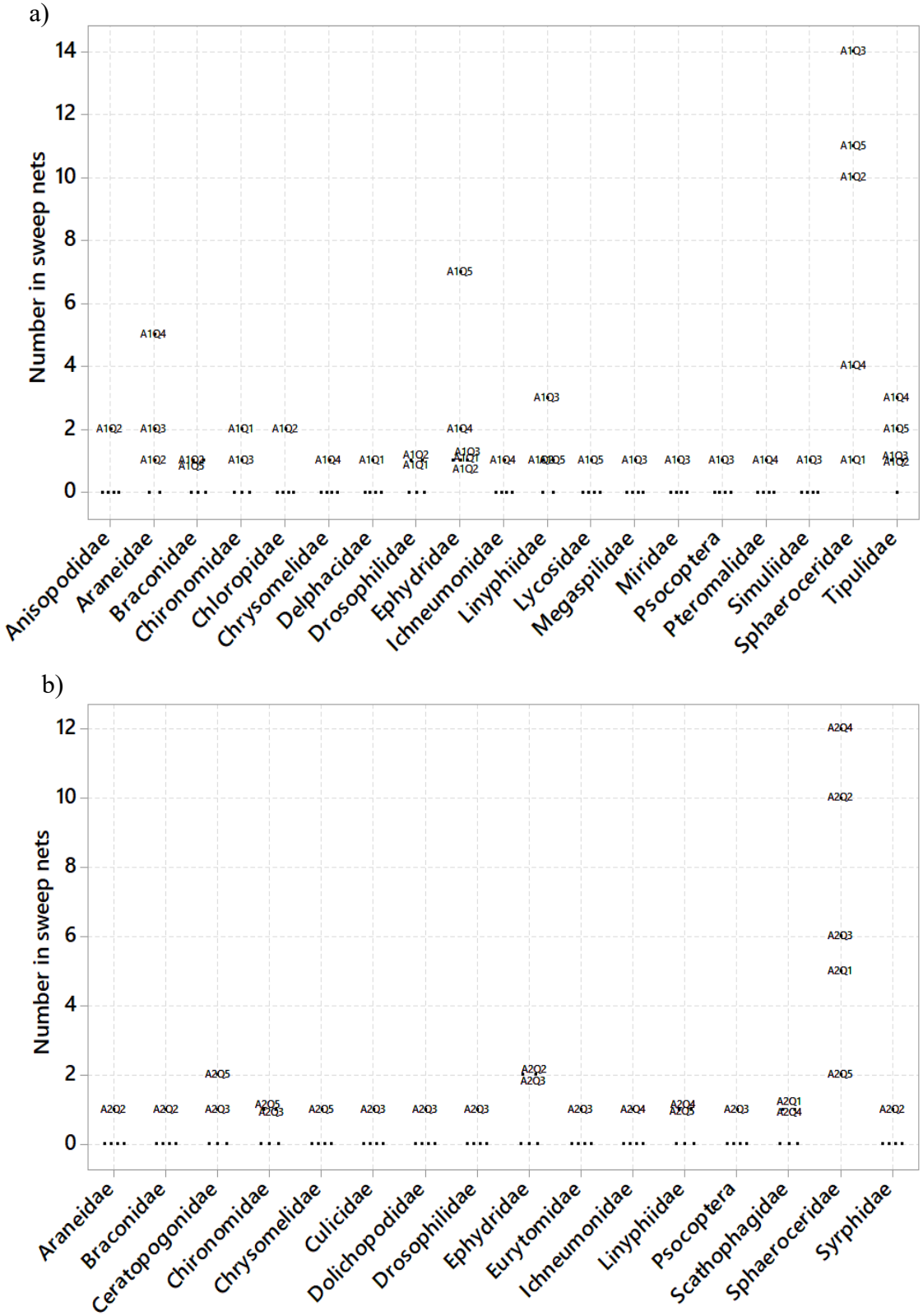


Figure 5.18 Number of invertebrates at family level caught in sweep nets in August 2015 on a) A1 (Q1-Q5; n = 5) and b) A2 (Q1-Q5; n = 5).

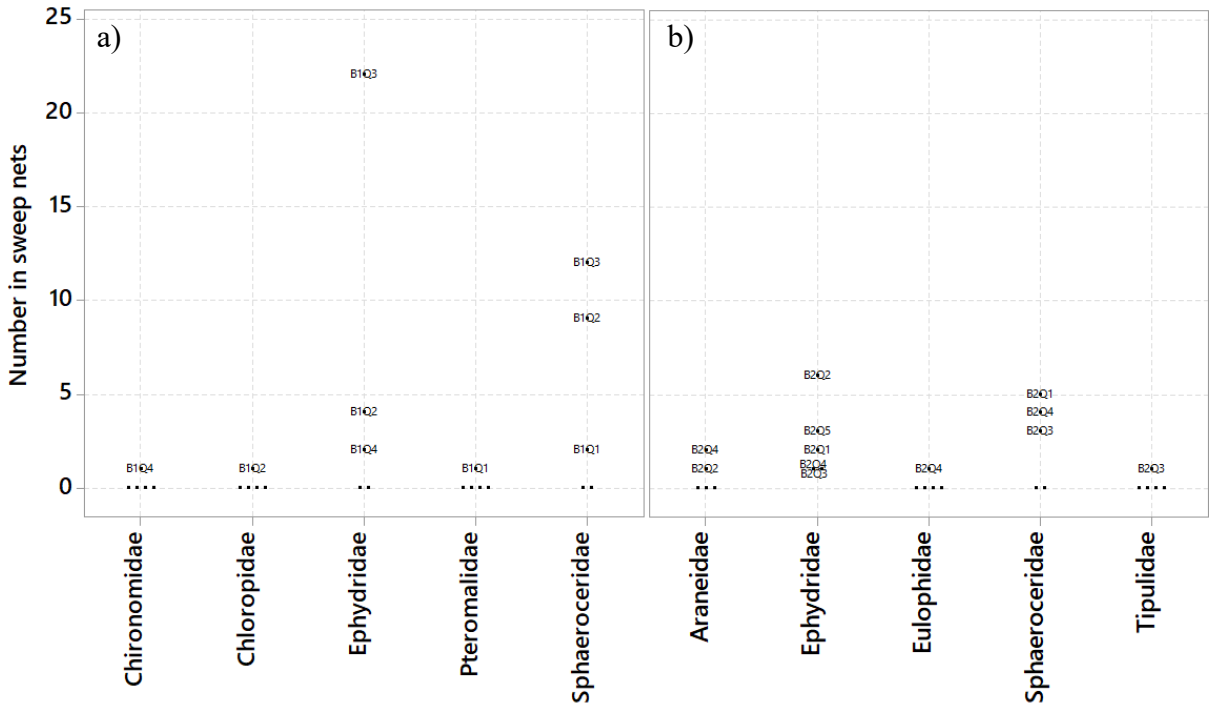


Figure 5.19 Number of invertebrates at family level caught in sweep nets in August 2015 on a) B1 (Q1-Q5; n = 5) and b) B2 (Q1-Q5; n = 5).

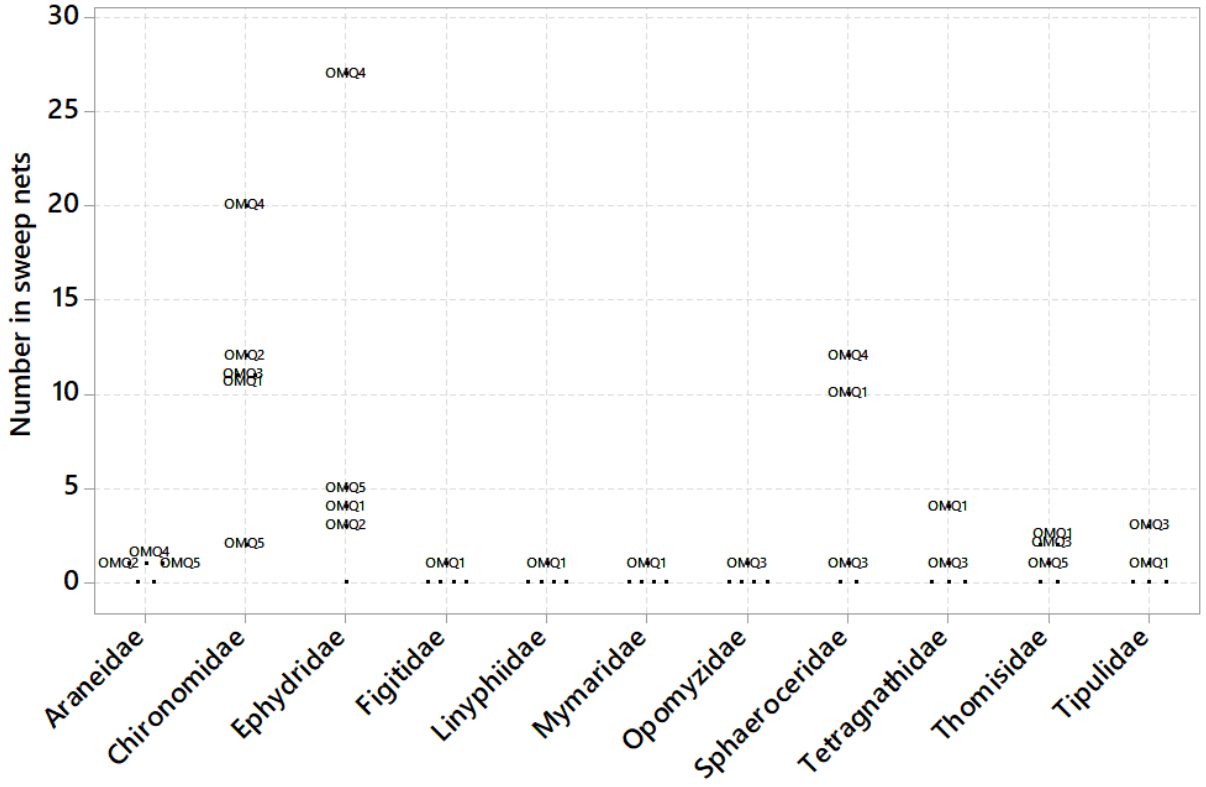


Figure 5.20 Number of invertebrates at family level caught in sweep nets in August 2015 on OM (Q1-Q5; n = 5).

5.3.2.4 Steart Marsh and Otterhampton Marsh August 2016 (post-breach)

A total of 164 specimens were caught in sweep nets on Steart Marsh and Otterhampton Marsh plots in August 2016, which is 78 less than were caught on the SSSI alone the same year. The plots with the most numerous specimens were A1 and A2 (≥ 50) and the highest number of family groups was found on A2, which is seven higher than on the SSSI (Table 5.15).

Table 5.15 Total invertebrate quantities and family groups caught in sweep nets on each study plot in August 2016. SSSI (collected in August 2016) is included for comparison.

	A1	A2	B1	B2	OM	SSSI
Total invertebrate quantity	50	71	10	33	0	242
Family group quantity	19	27	6	8	0	20

Arachnid numbers were very low or not present on the study plots, and all other specimens were insects. Specimens within the Tephritidae family were caught in greatest abundance on A1 and A2, although numbers were still low compared to the SSSI. This family was absent on all other plots (Table 5.16).

Specimens within the Tephritidae family were present in four out of five quadrats on A1 and in three out of five quadrats in A2, with the highest numbers being in A1Q3 on plot A1, and A2Q5 on plot A2 (Figure 5.21a).

The Site B plots had lower numbers of individual invertebrate specimens and family groups than the Site A plots. Of the families caught on the Site B plots, Chloropidae flies were the most numerous, and the highest numbers were caught in B1Q3 on plot B1 (Figure 5.22a) and B2Q2 and B2Q4 on Plot B2 (Figure 5.22b).

The plant coverage was too low on OM (70-100% bare ground) to take samples using sweep nets, so invertebrate numbers were counted as zero.

Table 5.16 Invertebrate families caught in sweep-nets on Steart Marsh study plots (A1-B2) and Otterhampton Marsh in August 2016.

Class	Order	Family	A1 QTY	A2 QTY	B1 QTY	B2 QTY	OM QTY
Arachnida	Araneae	Araneidae	1	4	0	0	0
		Linyphiidae	1	2	0	0	0
Insecta	Coleoptera	Chrysomelidae	1	0	0	0	0
		Coccinellidae	2	2	1	0	0
		Staphylinidae	0	0	0	1	0
	Diptera	Bibionidae	0	1	0	3	0
		Ceratopogonidae	5	10	0	0	0
		Chironomidae	3	3	1	19	0
		Chloropidae	4	3	4	0	0
		Drosophilidae	0	0	0	4	0
		Ephydridae	1	1	2	0	0
		Opomyzidae	0	0	0	1	0
		Phoridae	1	1	0	0	0
		Psilidae	1	1	0	0	0
		Scathophagidae	6	7	0	0	0
		Scatopsidae	0	1	0	2	0
		Sphaeroceridae	1	12	0	0	0
		Syrphidae	1	1	0	0	0
		Tephritidae	13	5	0	0	0
		Tipulidae	2	3	0	0	0
	Hemiptera	Cixiidae	2	2	0	1	0
		Delphacidae	0	1	0	0	0
		Miridae	2	1	1	0	0
		Psyllidae	0	1	0	0	0
	Hymenoptera	Braconidae	2	2	0	2	0
		Eulophidae	0	1	1	0	0
		Eurytomidae	0	1	0	0	0
		Figitidae	0	1	0	0	0
		Ichneumonidae	1	1	0	0	0
		Pteromalidae	0	2	0	0	0
	Psocoptera		0	1	0	0	0

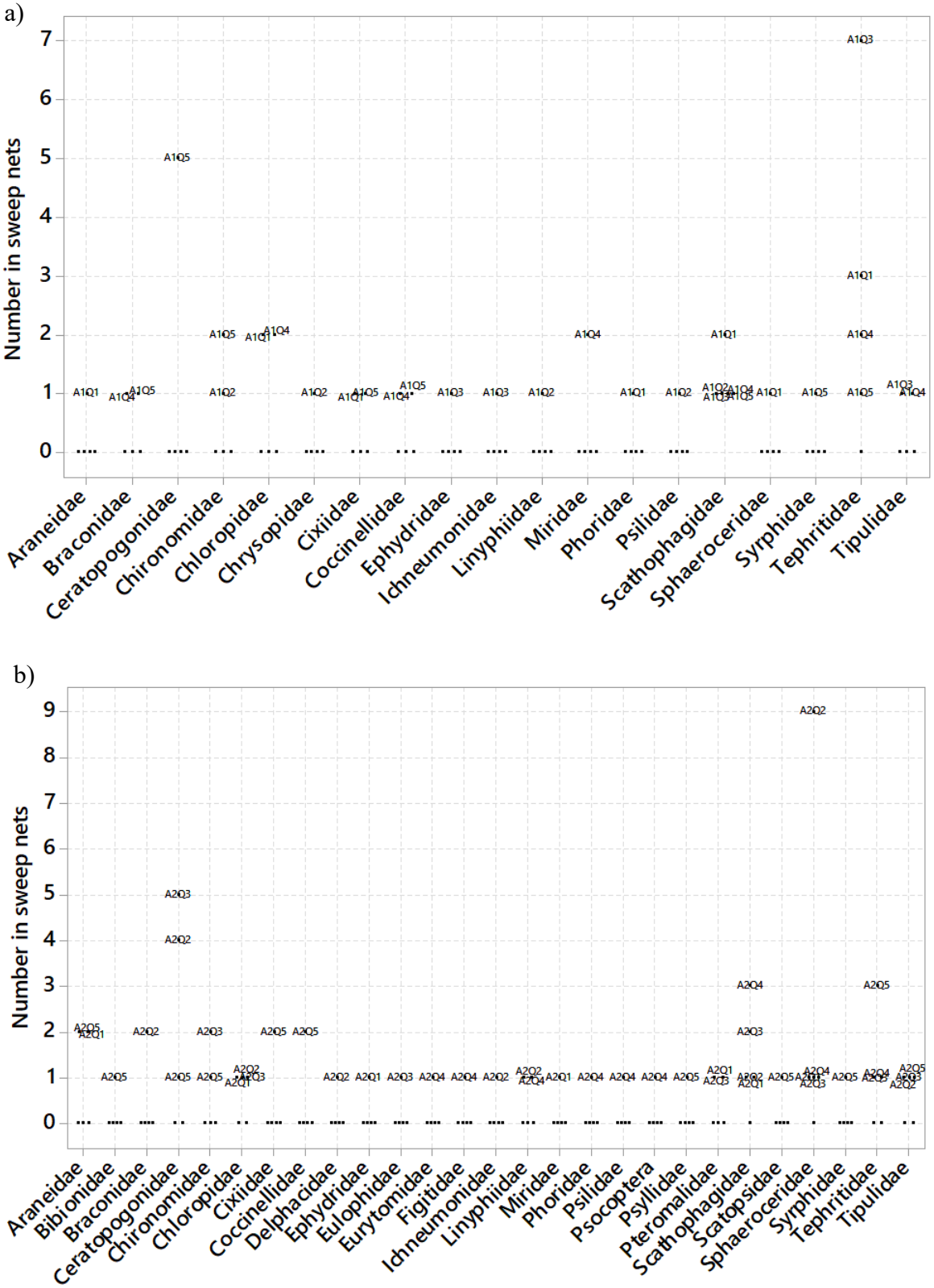


Figure 5.21 Number of invertebrates at family level caught in sweep nets in August 2016 on a) A1 (Q1-Q5; n = 5) and b) A2 (Q1-Q5; n = 5).

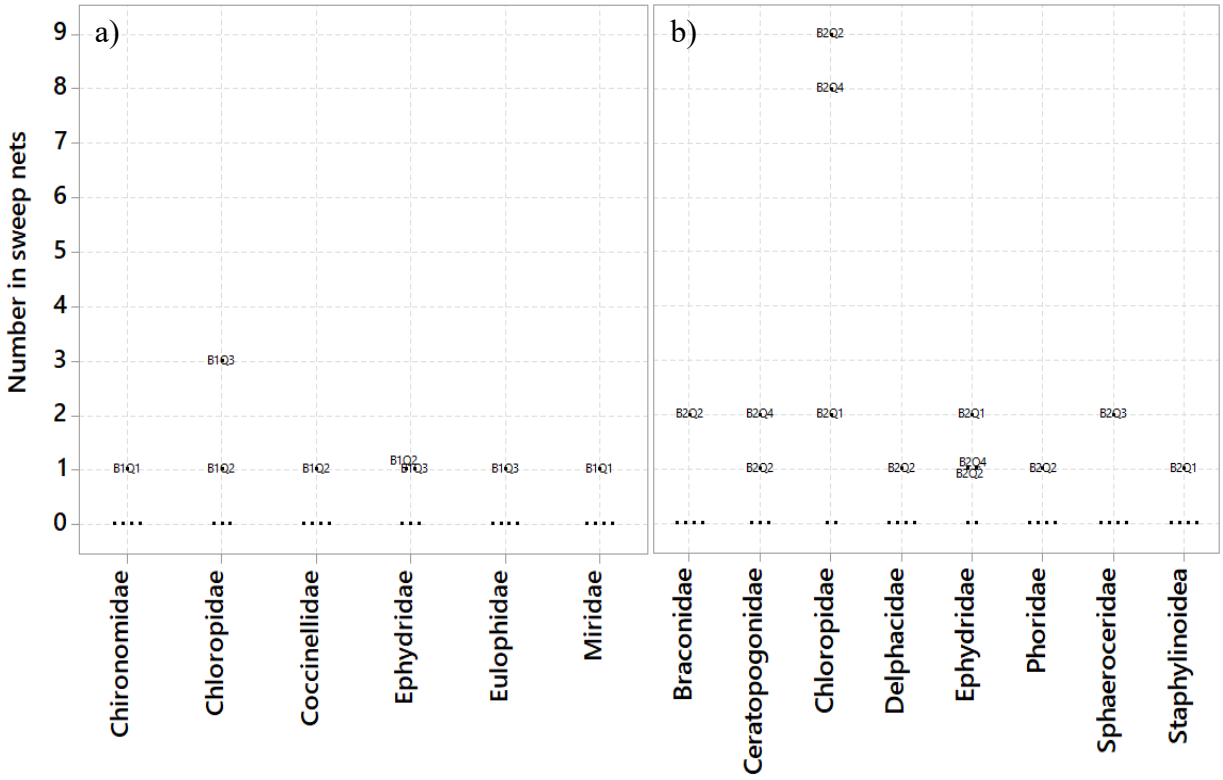


Figure 5.22 Number of invertebrates at family level caught in sweep nets in August 2016 on a) B1 (Q1-Q5; n = 5) and b) B2 (Q1-Q5; n = 5).

5.3.2.5 Steart Marsh and Otterhampton Marsh August 2017 (post-breach)

On most plots, numbers of plant-dwelling invertebrates caught in sweep nets was substantially higher in August 2017 than in 2016. Eight hundred and fifty-nine individual invertebrates were caught across all plots on Steart Marsh and OM. Higher numbers were caught on both A1 and A2 plots than were caught on the SSSI in August 2016. The number of families had dropped on A1 and A2, but numbers within these families was generally higher. A higher number of families were caught on B1 and B2 than the previous year. Invertebrate numbers on OM remained low, with 11 individuals caught within seven different families (Table 5.17).

Table 5.17 Total invertebrate quantities and family groups caught in sweep nets on each study plot in August 2017. SSSI (collected in August 2016) is included for comparison.

	A1	A2	B1	B2	OM	SSSI
Total invertebrate quantity	256	312	136	144	11	242
Family group quantity	16	17	14	14	7	20

The number of Tephritidae flies was much higher on plot A1 than the previous year, and 22 individuals were caught in A2. Specimens within this family were not caught on the Site B plots or OM.

Delphacidae numbers had increased substantially by August 2017 and there were numbers > 100 on A1 and > 150 on A2. Quantities were also > 50 on B1 and > 75 on B2. Only two Delphacidae specimens were caught on OM. Sphaeroceridae were also high in number on the Site A plots, and Miridae were found in all plots apart from OM, with the highest quantities in B2 (Table 5.18).

Table 5.18 Invertebrate families caught in sweep-nets on Steart Marsh study plots (A1-B2) and Otterhampton Marsh in August 2017.

Class	Order	Family	A1 QTY	A2 QTY	B1 QTY	B2 QTY	OM QTY
Arachnida	Araneae	Araneidae	1	3	2	2	0
		Linyphiidae	0	0	0	0	1
Insecta	Diptera	Anisopodidae	0	1	0	0	0
		Bibionidae	1	21	1	11	1
		Calliphoridae	1	0	0	0	0
		Ceratopogonidae	17	20	5	6	0
		Chironomidae	0	2	0	0	3
		Chloropidae	18	7	4	6	0
		Dolichopodidae	5	3	1	0	0
		Drosophilidae	0	2	0	0	0
		Ephydridae	0	0	2	2	1
		Lonchaeidae	0	0	0	1	0
		Micropezidae	0	0	0	1	0
		Psilidae	1	1	0	0	0
		Scathophagidae	7	6	2	1	0
		Sphaeroceridae	32	34	11	9	2
		Stratiomyidae	0	1	0	1	0
		Tephritidae	47	22	0	0	0
		Tipulidae	0	0	1	1	1
	Hemiptera	Anthocoridae	0	1	0	0	0
		Aphididae	4	10	1	0	0
		Cicadellidae	6	0	1	2	0
		Delphacidae	102	152	56	77	2
		Miridae	11	26	46	24	0
	Hymenoptera	Braconidae	1	0	0	0	0
		Ichneumonidae	2	0	0	0	0
		Pteromalidae	0	0	1	0	0

Tephritidae flies were found in all quadrats on plot A1 with the highest numbers in A1Q3 (Figure 5.23a). Specimens of this family were caught in four of the five quadrats on A2 and were absent in A2Q3 and most abundant in A2Q4 (Figure 5.23b). Delphacidae were

found in three of the five quadrats on A1, and numbers were highest in A1Q4 (> 75 individuals). They were caught in four of the five quadrats on A2, with numbers being highest in A2Q5 (> 100 individuals). Delphacidae were caught in four of five quadrats in B1 (Figure 5.24a) and in all quadrats on B2 (Figure 5.24b), although numbers were lower overall than on the Site A plots. The two Delphacidae specimens caught on OM were found in OMQ4, which was the only quadrat on OM to contain plant-dwelling invertebrates in August 2017 (Figure 5.25).

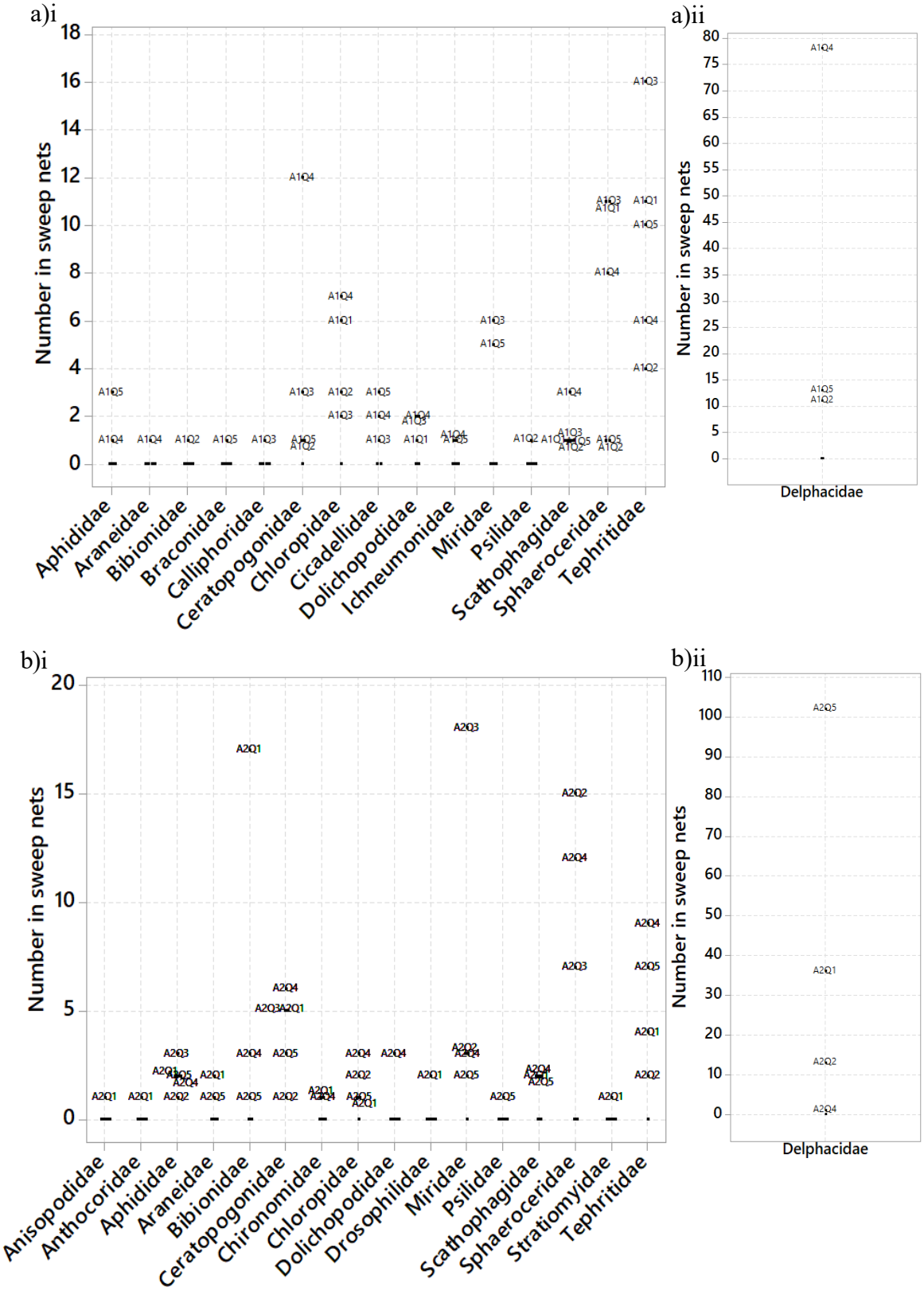


Figure 5.23 Number of invertebrates at family level caught in sweep nets in August 2017 on a)i A1 (Q1-Q5; n = 5) and b)i A2 (Q1-Q5; n = 5); a)ii is Delphacidae caught in August 2017 on A1; b)ii is Delphacidae caught in August 2017 on A2.

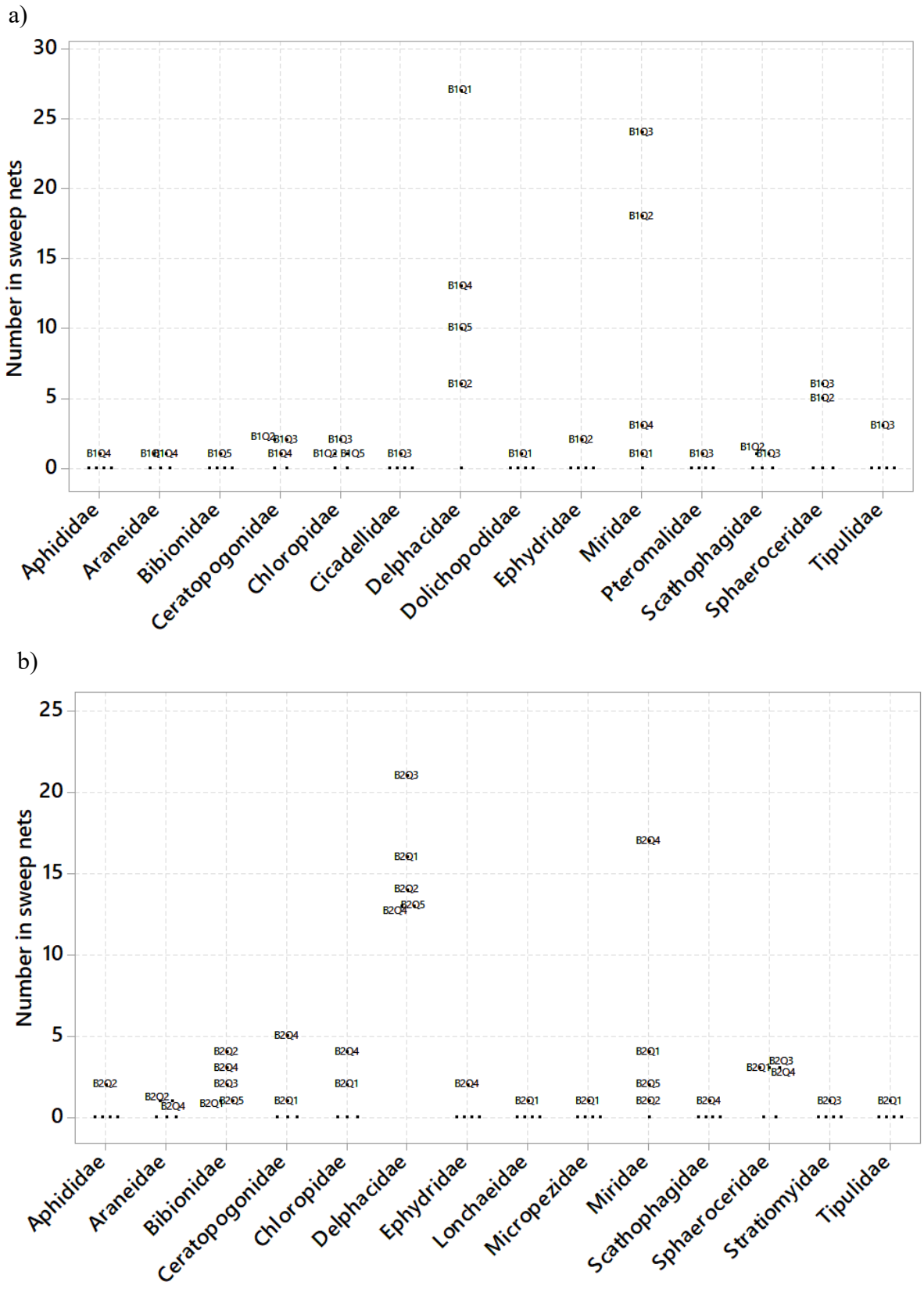


Figure 5.24 Number of invertebrates at family level caught in sweep nets in August 2017 on a) B1 (Q1-Q5; n = 5) and b) B2 (Q1-Q5; n = 5).

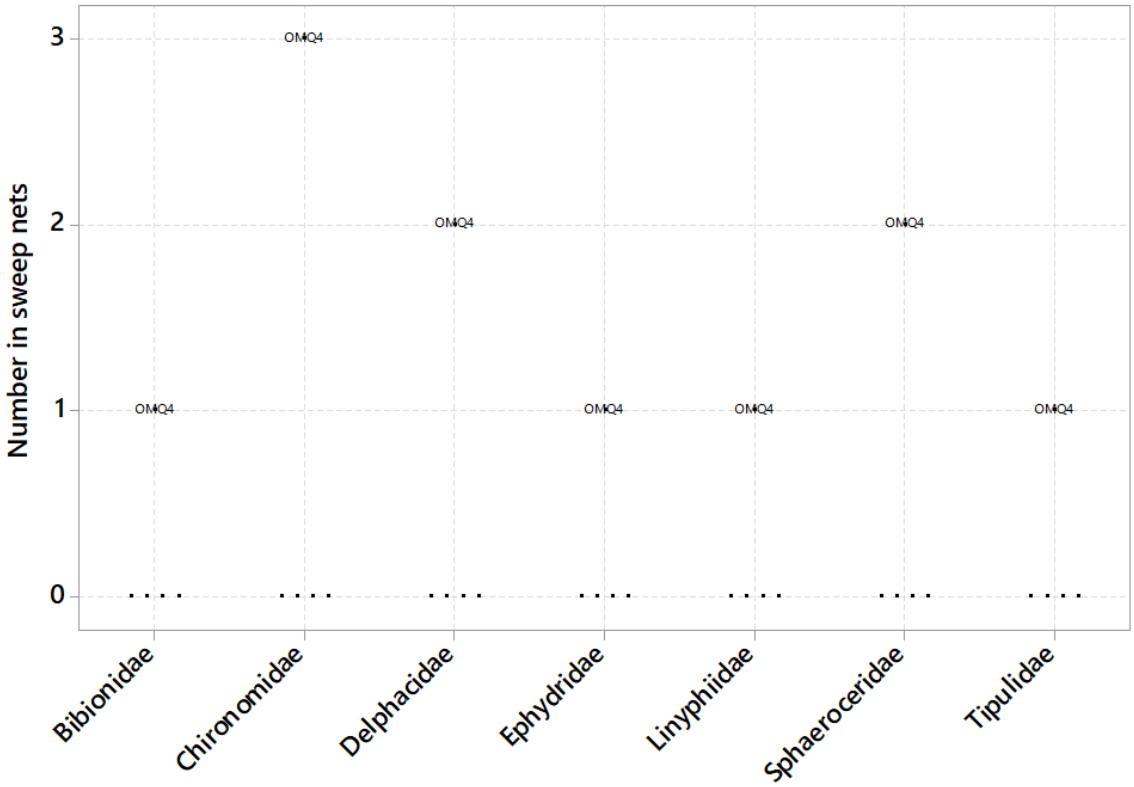


Figure 5.25 Number of invertebrates at family level caught in sweep nets in August 2017 on OM (Q1-Q5; n = 5).

5.3.3 Cluster analysis and ordination

5.3.3.1 Ground-dwelling invertebrates

5.3.3.1.1 August 2014 (pre-breach)

Clustering indicates that the SSSI quadrats differed from those on Steart Marsh and OM in terms of invertebrate families caught in pitfall traps in August 2014 (Figure 5.26). This is also evidenced in DCA ordination which shows that the former agricultural sites differed from the SSSI (Figure 5.27). This was expected due to the different starting states across all sites.

When analysed with 2014 data, cluster analysis (Figure 5.26) shows that SSSI Q1 and SSSI Q2 were different to the other quadrats in terms of ground-dwelling invertebrates. This was due to the abundance of Talitridae in these quadrats, which was disproportionately higher than in the other three quadrats.

Some families, such as Lycosidae spiders were found on both agricultural land and the saltmarsh. Wolf spiders are found in many different systems, but most cannot tolerate saltmarsh habitats, so the species within this family were not the same. Members of the *Pardosa* genus were found in both habitats, but the SSSI species was likely to be *P.*

purbeckensis, although it is difficult to discern between this species and *P. agrestis* that is found on dry land.

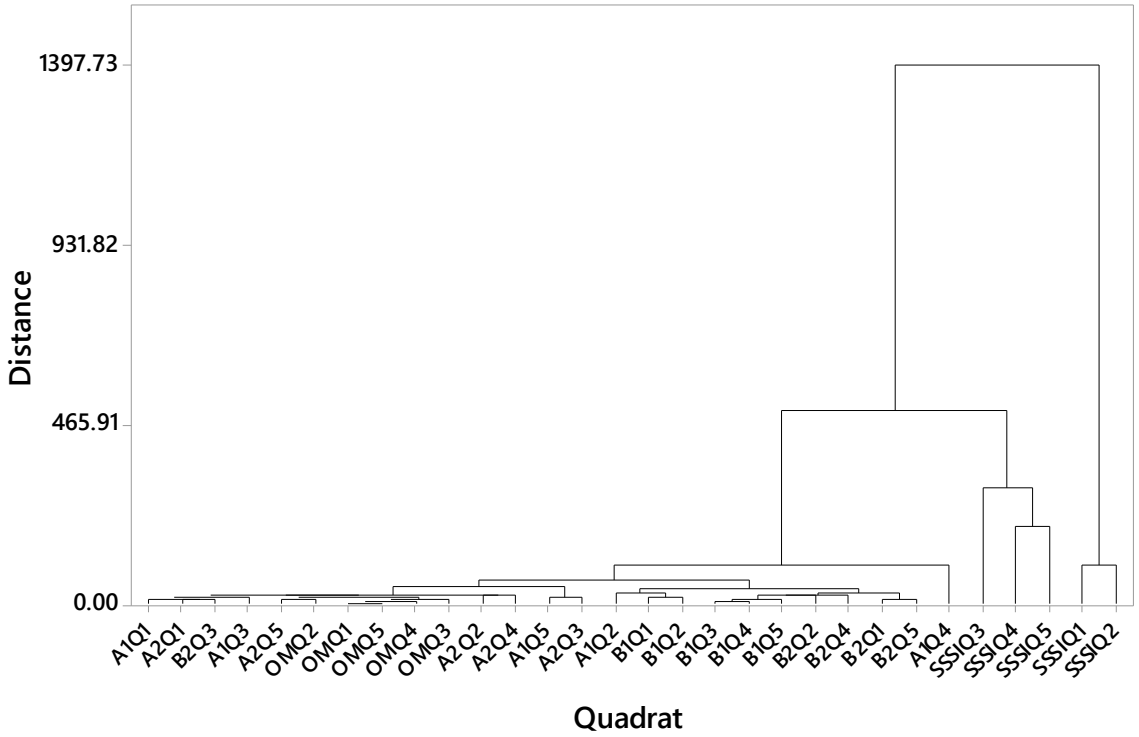


Figure 5.26 Clustering of study site quadrats using observations of number of individual invertebrates recorded at family level caught in pitfall traps in August 2014. SSSI data are from August 2016. Average linkage and Euclidean distance were used to calculate clusters.

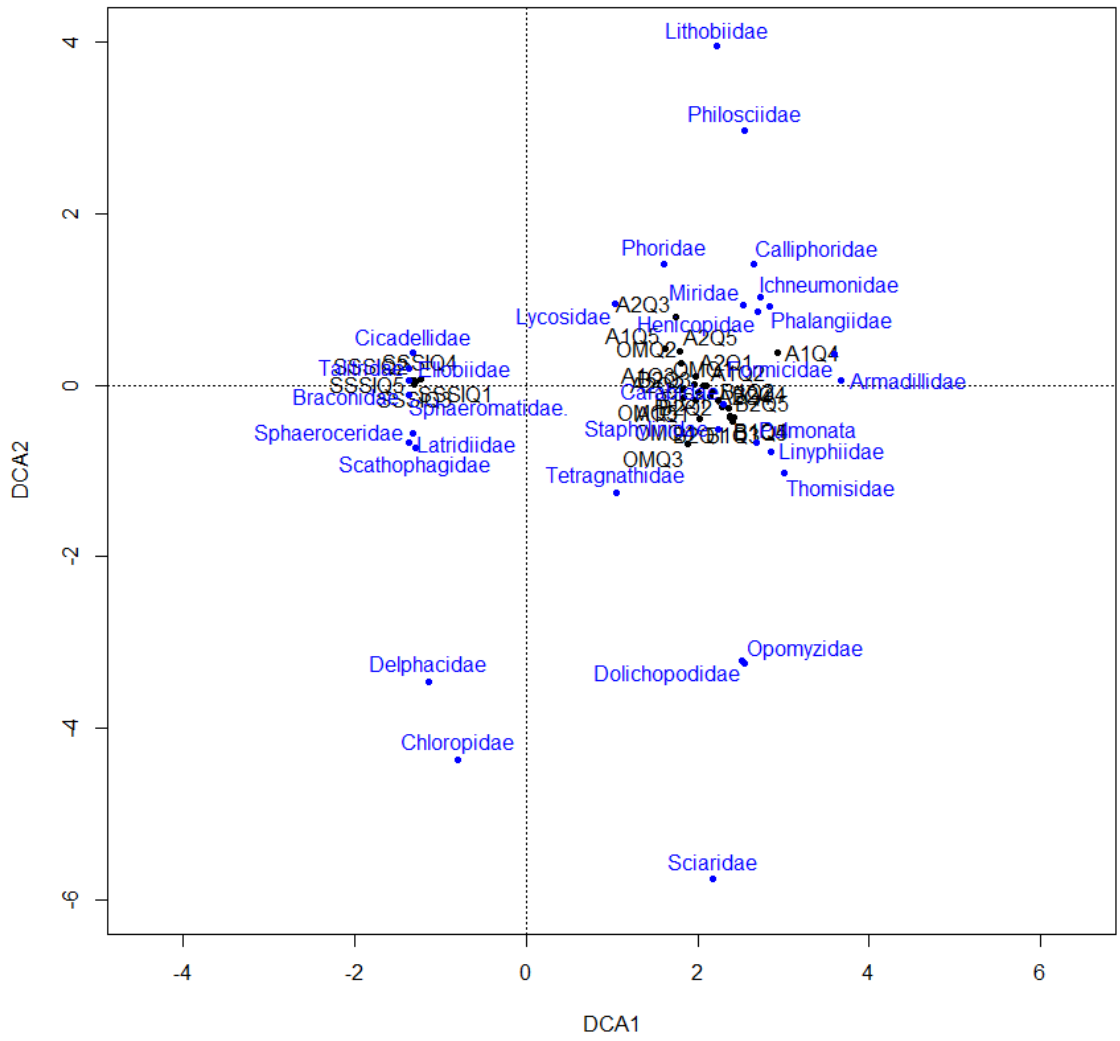


Figure 5.27 DCA ordination of invertebrates caught in pitfall traps on all study plots in Aug 2014. SSSI data are from August 2016.

5.3.3.1.2 August 2015 (one-year post-breach)

Numbers of ground-dwelling invertebrates were low in 2015 across all plots, which is reflected in the cluster analysis (Figure 5.28). This shows similarities of the Steart Marsh and OM quadrats, which were very different to the SSSI where ground-dwelling invertebrates were numerous. Ordination reflects this difference and shows that family groupings were still mostly different in 2015 (Figure 5.29).

Ordination shows that some Site A quadrats were moving toward the SSSI in August 2015. A1Q3 and A2Q4 were the most similar to the SSSI at this time, although the differences were still substantial due to the high number of amphipods on the SSSI.

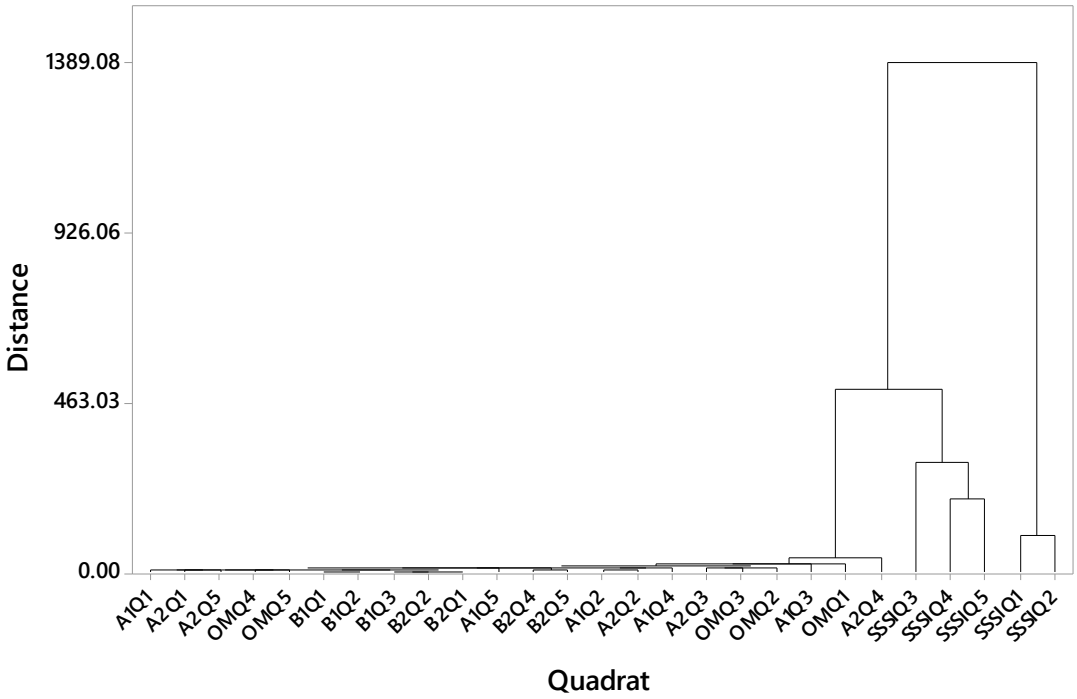


Figure 5.28 Clustering of study site quadrats using observations of number of individual invertebrates recorded at family level caught in pitfall traps in August 2015. SSSI data are from August 2016. Average linkage and Euclidean distance were used to calculate clusters.

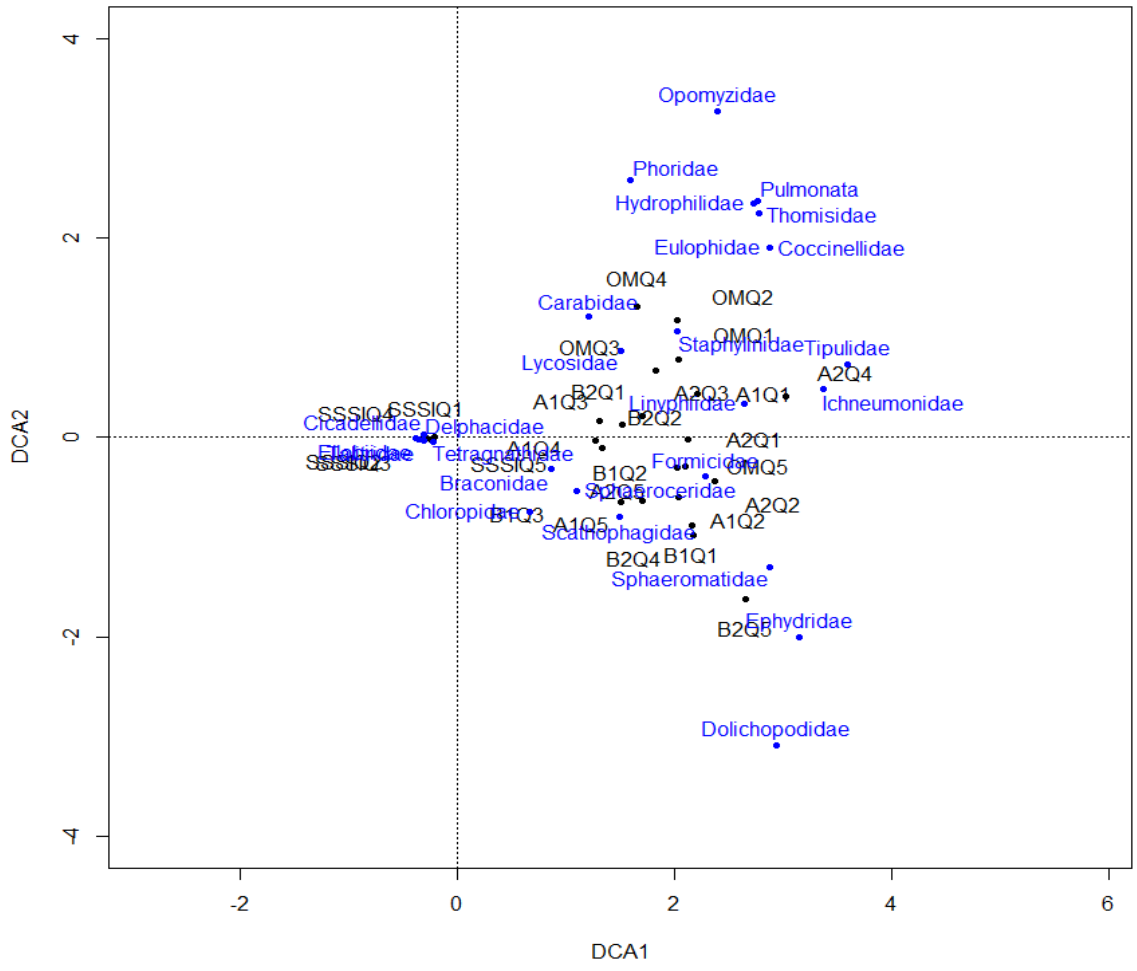


Figure 5.29 DCA ordination of invertebrates caught in pitfall traps on all study plots in August 2015. SSSI data are from August 2016.

5.3.3.1.3 August 2016 (two years post-breach)

Although quadrats on Steart Marsh and OM were still different from the SSSI in 2016, some of the Site A quadrats were more similar to SSSI quadrats than they had been the previous year, although the number of amphipods caught in SSSIQ1 and SSSIQ2 meant that these quadrats were still a large distance from the Site A quadrats (Figure 5.30).

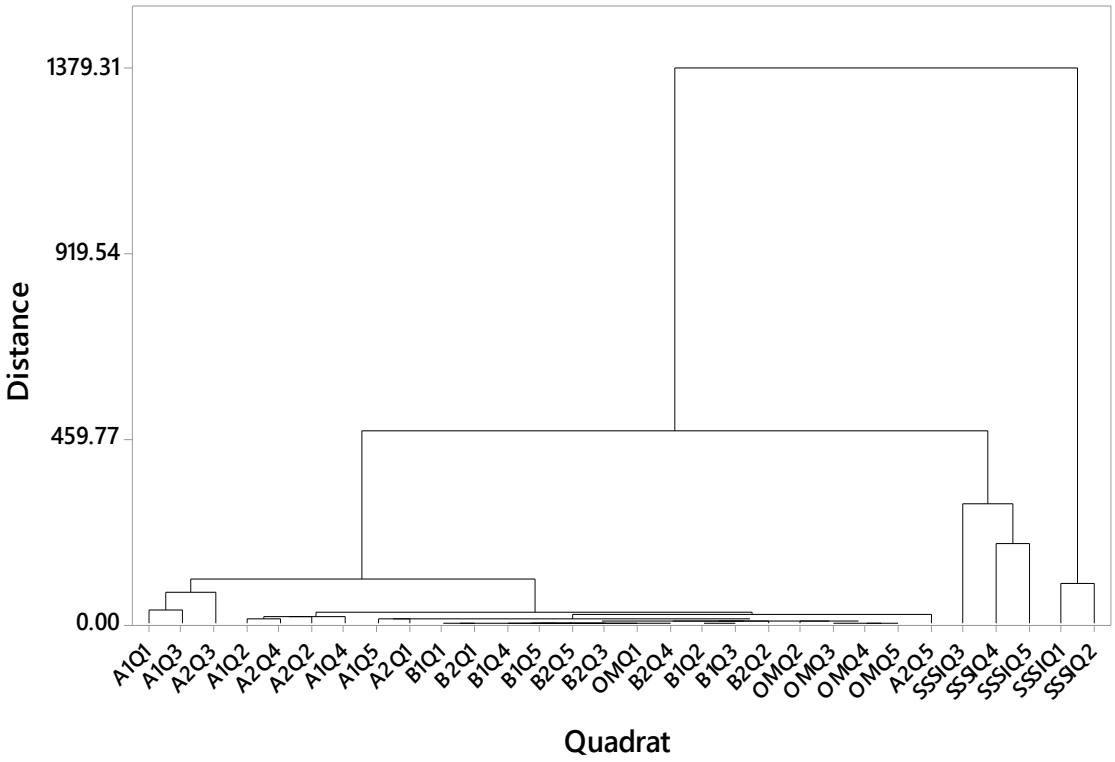


Figure 5.30 Clustering of study site quadrats using observations of number of individual invertebrates recorded at family level caught in pitfall traps in August 2016. Average linkage and Euclidean distance were used to calculate clusters.

Quadrats A1Q1, A1Q3 and A2Q3 were grouped together in cluster analysis, and ordination (Figure 5.31) shows that these quadrats had scores closer to the SSSI quadrats. This occurred because Talitridae amphipods were beginning to colonise these plots, and they were caught in high numbers in these quadrats (see section 5.3.1.4). Other Site A quadrats were becoming more similar to these quadrats and the SSSI, although they were still considerably different than SSSIQ1 and SSSIQ2. Some Site A, Site B, and OM quadrats were still similar in August 2016.

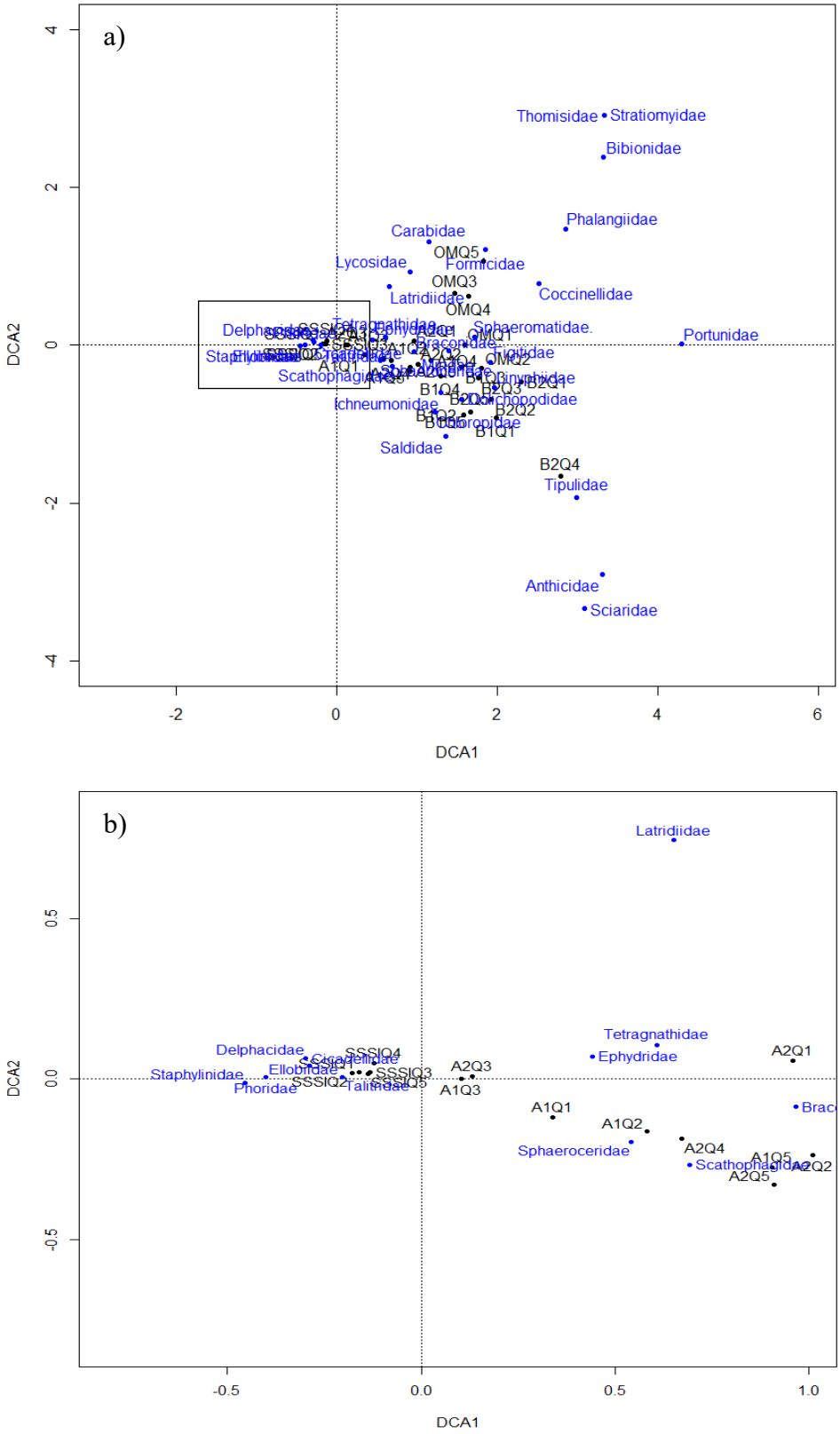


Figure 5.31 DCA plots of quadrats grouped by invertebrate families caught in pitfall traps in August 2016; a) complete biplot of quadrats and invertebrates responsible for groupings in August 2016; b) magnified plot showing groupings of the SSSI quadrats and quadrats that are closest to them in relation to invertebrate families. Black box in a) shows the area of the plot that is magnified in b).

5.3.3.1.4 August 2017 (three years post-breach)

Quadrats A1Q5 and A2Q4 were very close in distance to SSSIQ5 in August 2017 and they were the closest Steart Marsh quadrats to the SSSI (Figure 5.32). More than 200 Talitridae amphipods were caught in both quadrats, which is similar to numbers caught in SSSIQ5.

These quadrats are also close to the SSSI quadrats on the first axis of ordination (Figure 5.33). DCA1 is the only meaningful axis (> 0.5), and site scores show that A1Q4 was the closest in DCA analysis. This is mainly influenced by Talitridae, Ellobiidae and Sphaeroceridae.

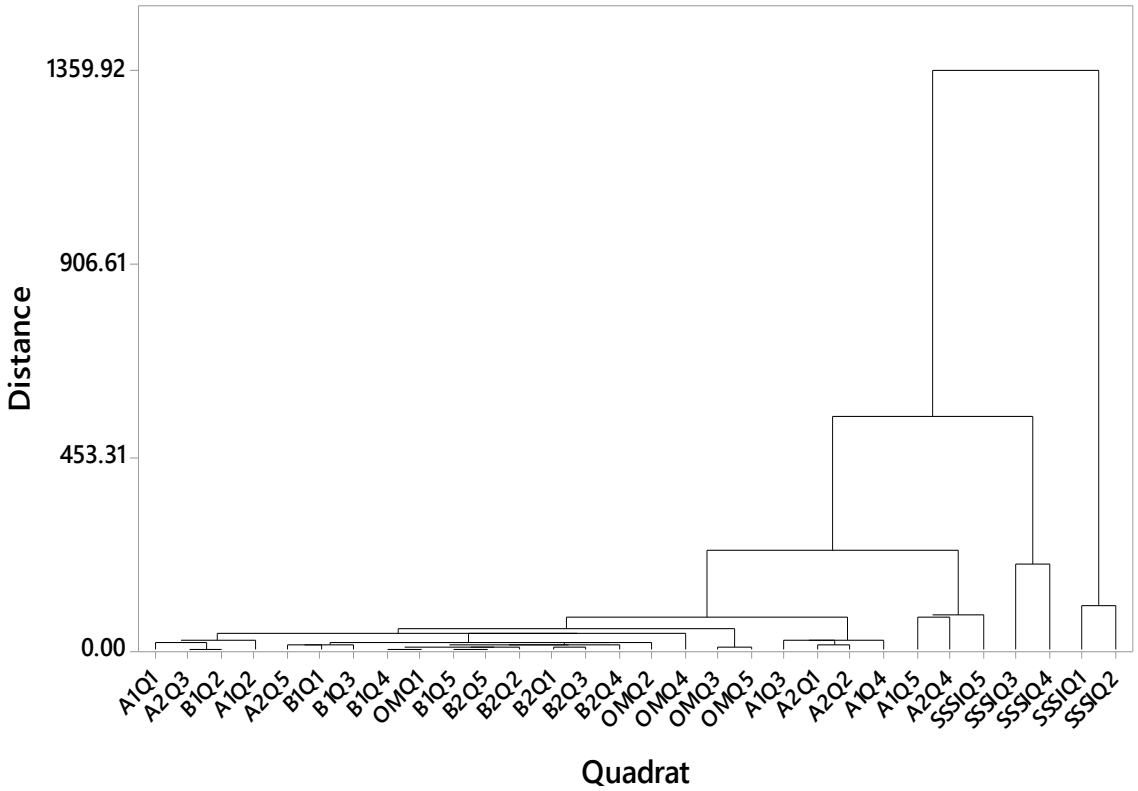


Figure 5.32 Clustering of study site quadrats using observations of number of individual invertebrates recorded at family level in pitfall traps in August 2017. SSSI data are from August 2016. Average linkage and Euclidean distance were used to calculate clusters.

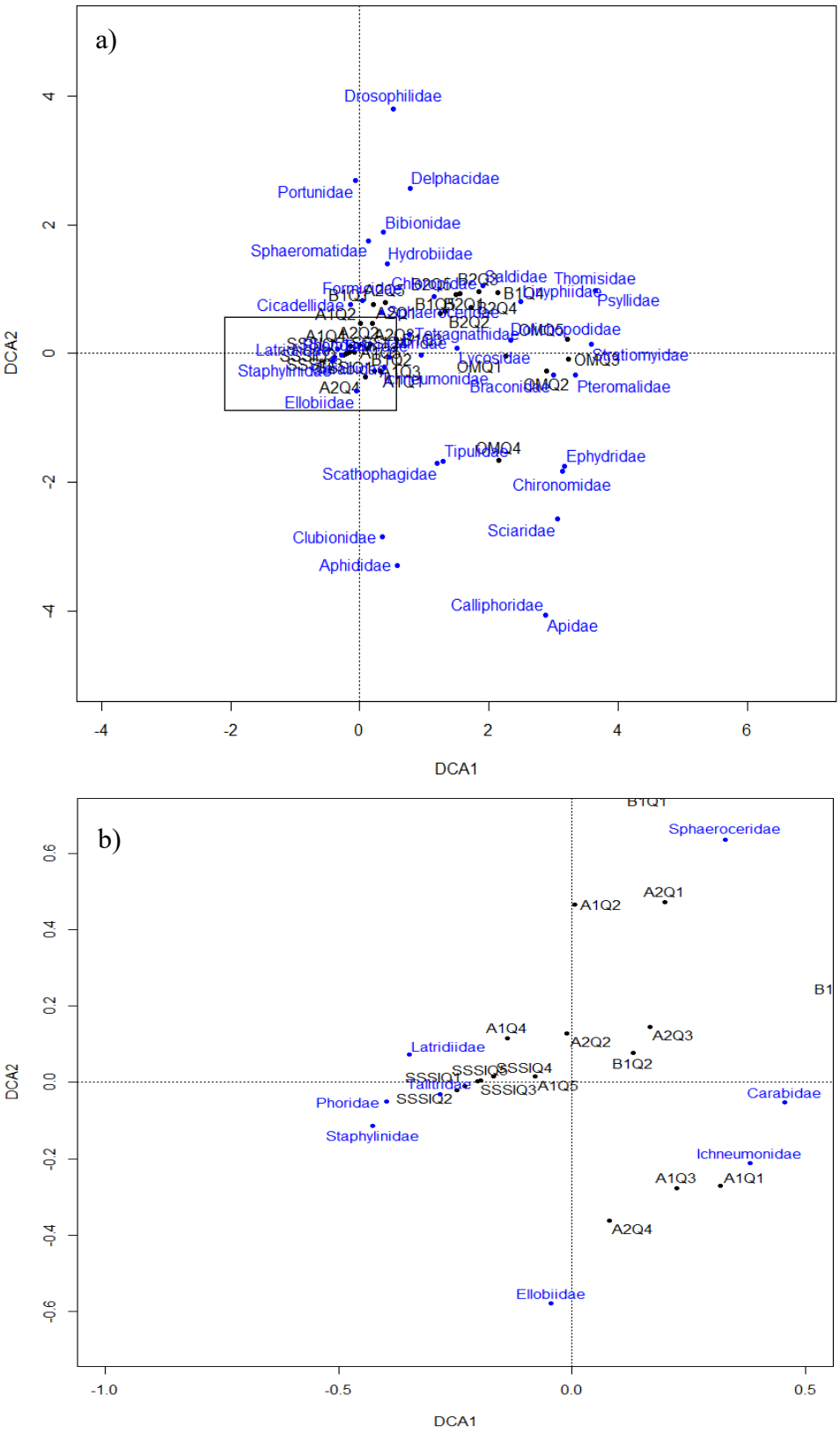


Figure 5.33 DCA plots of quadrats grouped by invertebrate families caught in pitfall traps in August 2017; a) complete biplot of quadrats and invertebrates responsible for groupings in August 2017; b) magnified plot showing groupings of the SSSI quadrats and quadrats that are closest to them in relation to invertebrate families. Black box in a) shows the area of the plot that is magnified in b).

Procrustes error plots show how ground-dwelling invertebrate assemblages changed from 2014-2017 (Figure 5.34). The Site A quadrats moved toward the SSSI quadrats, but most of the Site B quadrats did not move as far or moved in another direction. OM quadrats moved in the opposite direction, quite a distance from original starting positions. Procrustes residuals (Figure 5.35) showed that OM quadrats changed considerably, because they had different assemblages to their starting state, and they were very different from the other plots by 2017. A1Q4 changed the most out of all the Site A quadrats, with invertebrate families becoming similar to the SSSI in this quadrat, and to other Site A quadrats.

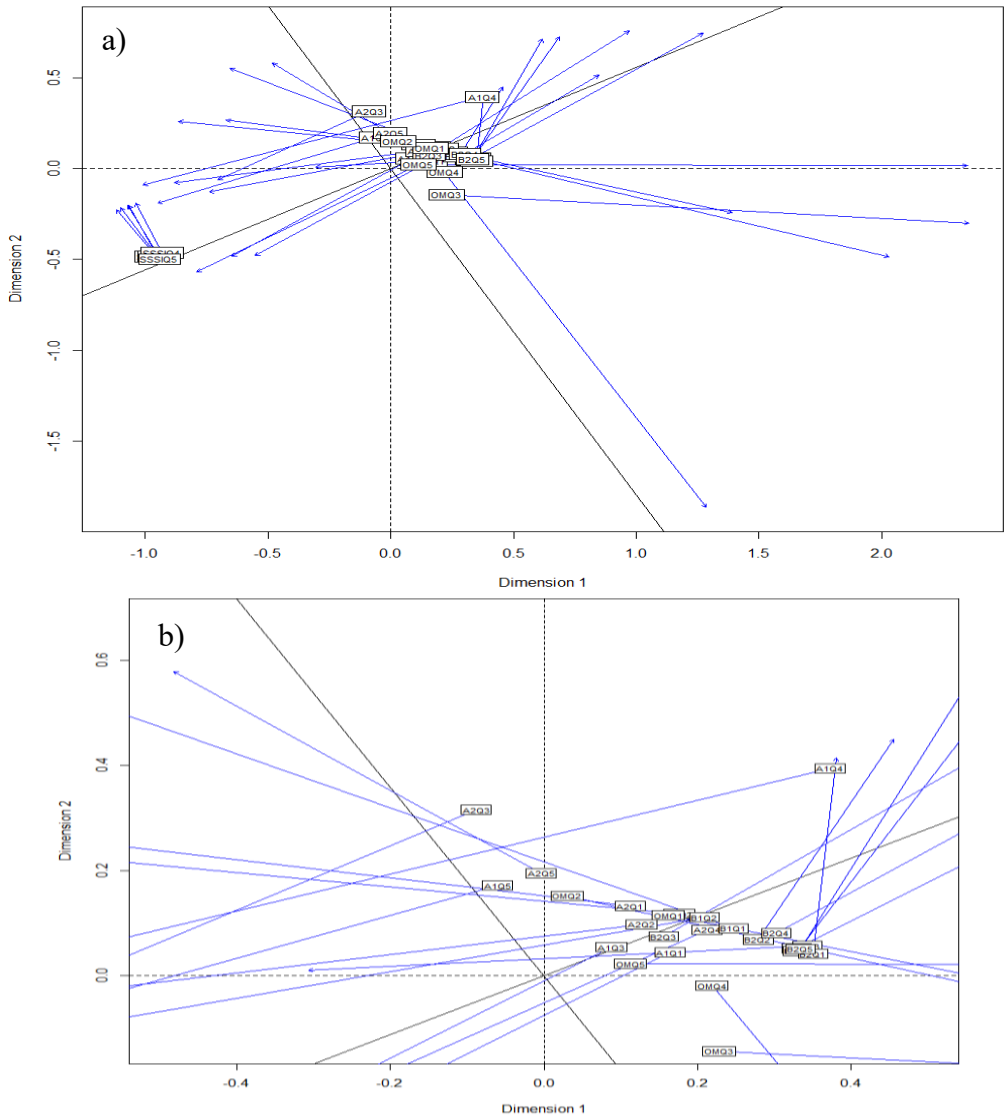


Figure 5.34 A comparison of DCA ordinations of ground-dwelling invertebrates in August 2014 and 2017; a) Procrustes plot of 2014 and 2017 ordinations; b) magnified image of the plot. Quadrat labels are the position of quadrats in 2014 and arrow heads are the position of quadrats in 2017. The SSSI data was the same in both ordinations (recorded in August 2016), but moved in relation to the other quadrats.

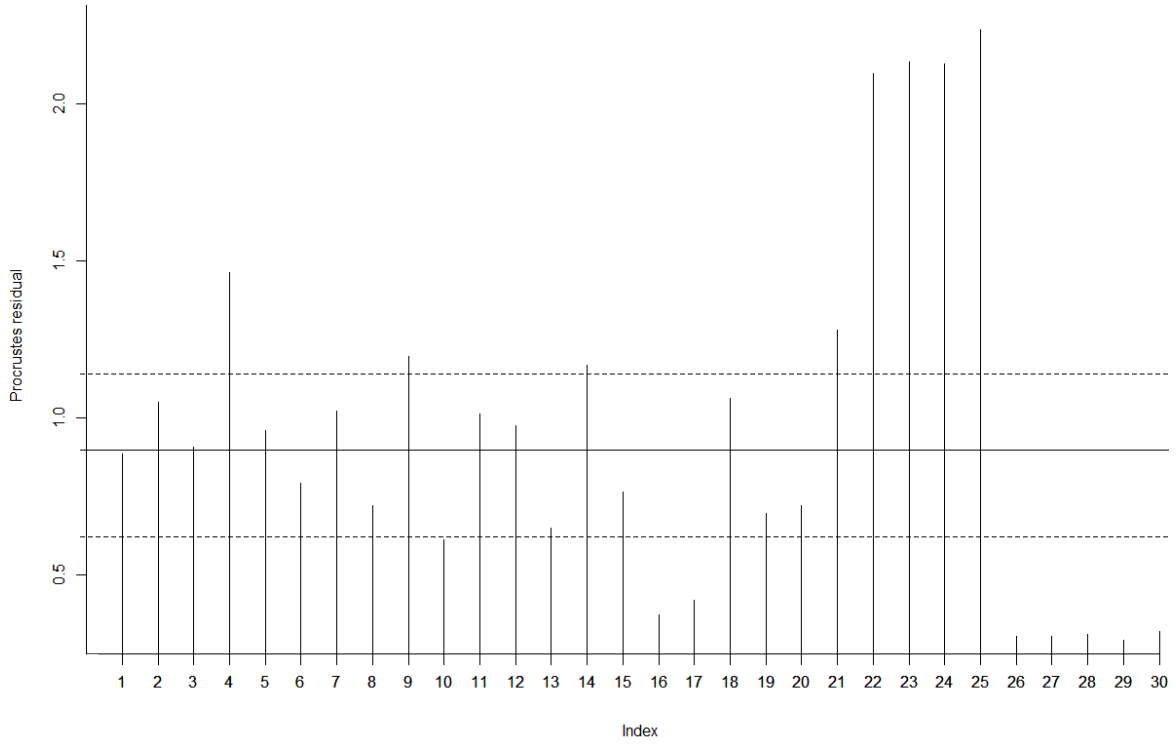


Figure 5.35 Procrustes error residuals for 2014 and 2017 ground-dwelling invertebrate ordinations. Index relates to quadrats: 1-5 = A1Q1-A1Q5; 6-10 = A2Q1-A2Q5; 11-15 = B1Q1-B1Q5; 16-20 = B2Q1-B2Q5; 21-25 = OMQ1-OMQ5; 26-30 = SSSIQ1-SSSIQ2. 25-75 % quantiles are between dotted lines and the solid line represents 50% quantiles of the residuals.

5.3.3.2 Plant-dwelling invertebrates

5.3.3.2.1 August 2014 (pre-breach)

SSSIQ2 was different than the other SSSI quadrats in terms of plant-dwelling invertebrates (Figure 5.36). Although Delphacidae were found in all quadrats on the SSSI they were particularly numerous in SSSIQ2.

The quadrats on Steart Marsh and OM were generally similar, although A2Q2 and A2Q4 were closer in distance to the SSSI than the other quadrats were. Delphacidae were found in these quadrats, which may be why there are some similarities, although they were not the same species as found on the SSSI (see section 5.3.5 for information on the saltmarsh Delphacidae species). Ordination shows that the SSSI quadrats were grouped together and the Steart Marsh and OM quadrats had very different scores to the SSSI based on families caught in sweep nets (Figure 5.37).

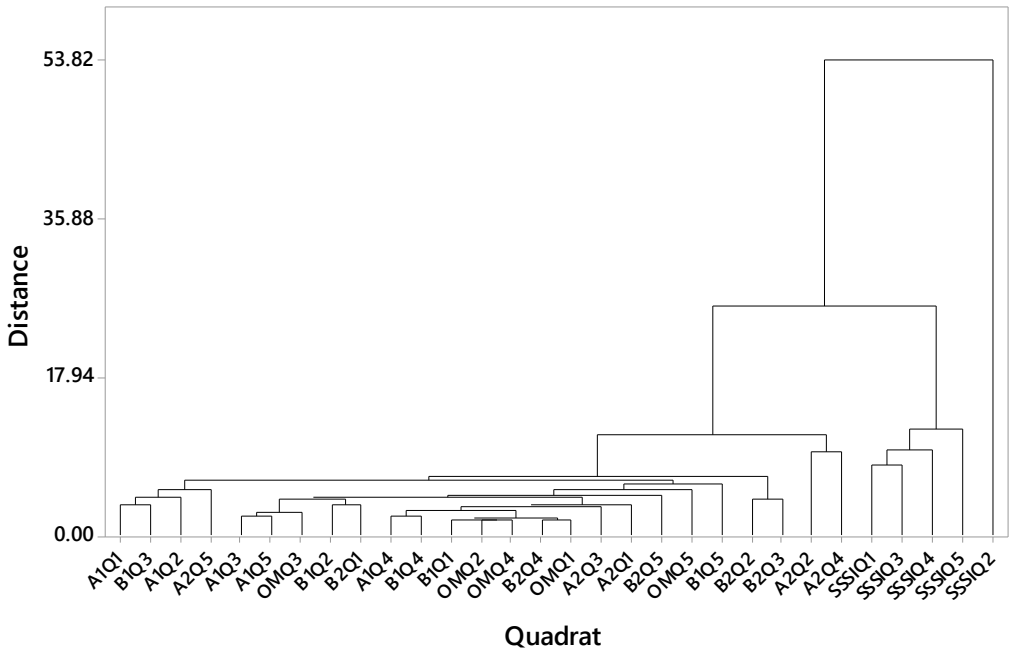


Figure 5.36 Clustering of study site quadrats using observations of number of individual invertebrates recorded at family level caught in sweep nets in August 2014. SSSI data are from August 2016. Average linkage and Euclidean distance were used to calculate clusters.

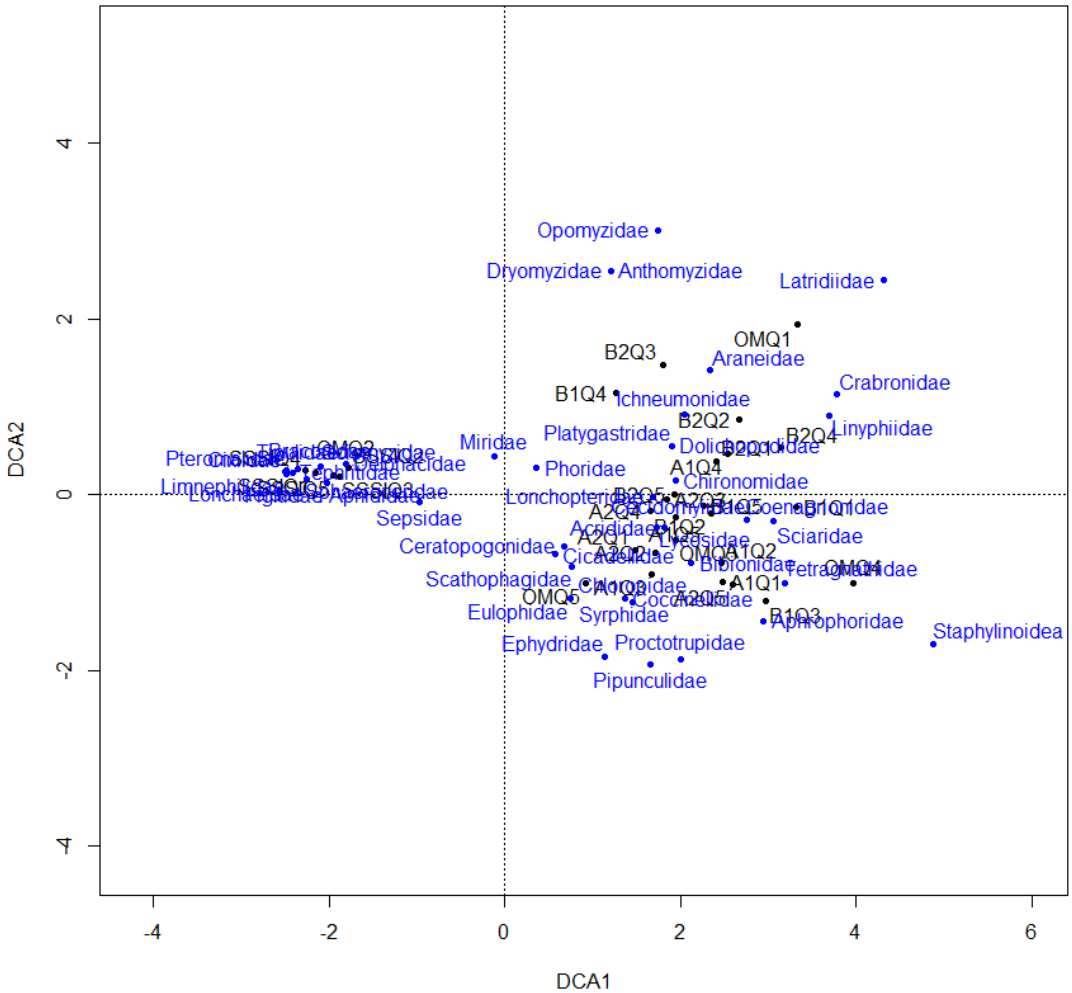


Figure 5.37 DCA ordination of invertebrates caught in sweep nets in Aug 2014 on all study plots.

5.3.3.2.2 August 2015 (one-year post-breach)

Cluster analysis shows that OMQ4 was different than the other quadrats on Steart Marsh and OM apart from B1Q3, which was primarily due to the number of Ephydridae caught in these quadrats (see section 5.3.1.3). The dendrogram (Figure 5.38) shows that these quadrats were closer to four of the SSSI quadrats than SSSIQ2 was. However, this is not clearly shown in the ordination (Figure 5.39), which shows that all SSSI quadrats are different in scores on the first axis than Steart Marsh and OM quadrats.

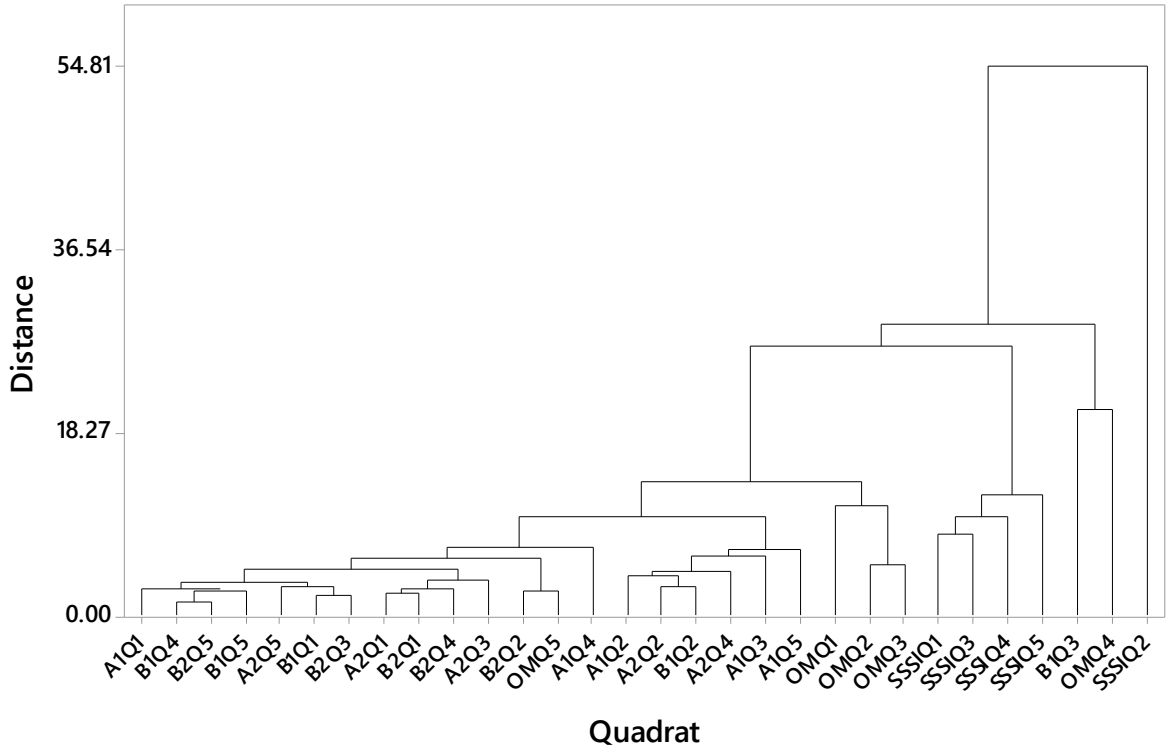


Figure 5.38 Clustering of study site quadrats using observations of number of individual invertebrates recorded at family level caught in sweep nets in August 2015. SSSI data are from August 2016. Average linkage and Euclidean distance were used to calculate clusters.

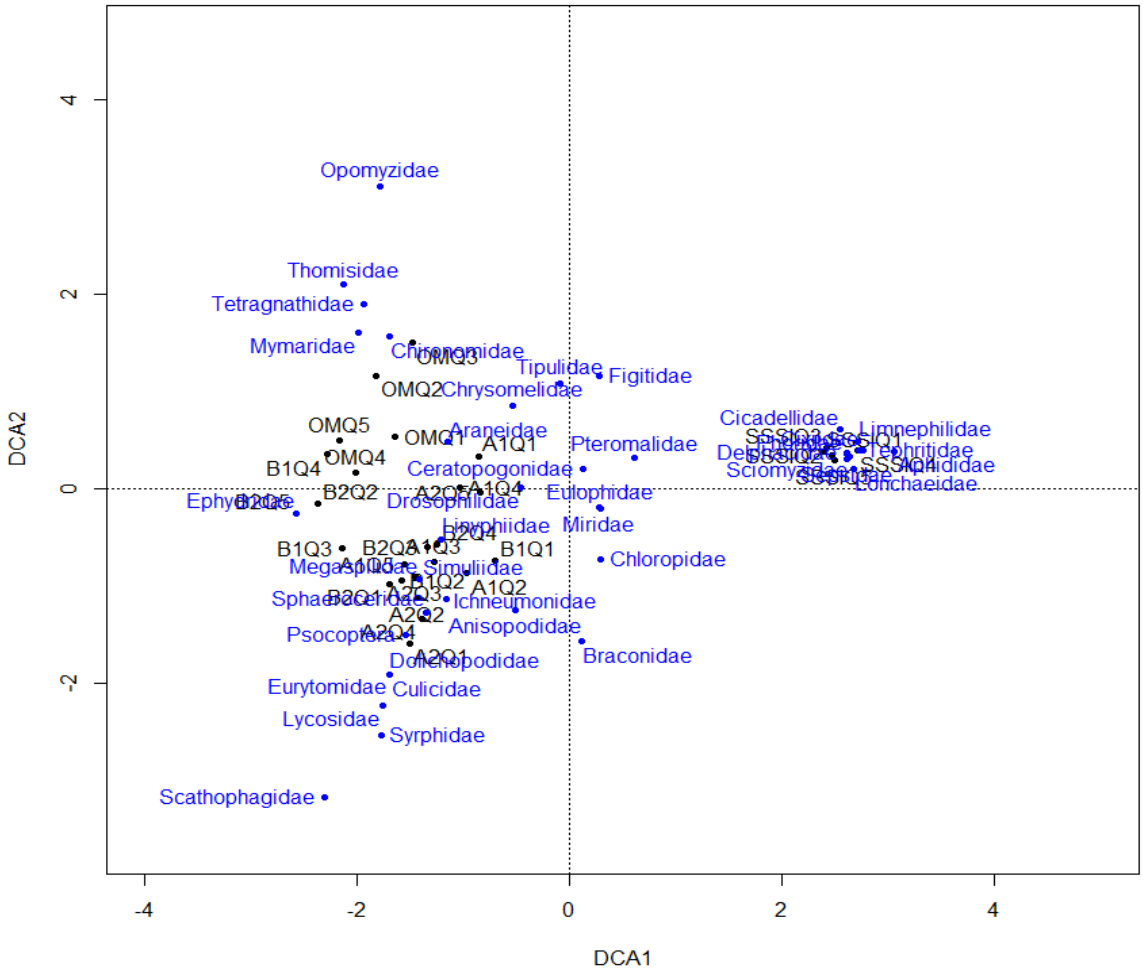


Figure 5.39 DCA ordination of invertebrates caught in sweep nets in Aug 2015 on all study plots. SSSI data from August 2016.

5.3.3.2.3 August 2016 (two years post-breach)

Cluster analysis (Figure 5.40) shows that A2Q2 was the closest in distance to SSSI quadrats in August 2016. This similarity is on the second axis on the ordination (Figure 5.41). Eigenvalues are > 0.5 on DCA1 and DCA2, so both are meaningful. A1Q3 was the quadrat closest in scores to the SSSI quadrats on the first and second axes on ordination and is closer in distance to the SSSI than 70% of other quadrats in cluster analysis. This similarity appears to be influenced by the relatively high number of Tephritidae in this quadrat.

Figure 5.40 Clustering of study site quadrats using observations of number of individual invertebrates recorded at family level caught in sweep nets in August 2016. Average linkage and Euclidean distance were used to calculate clusters.

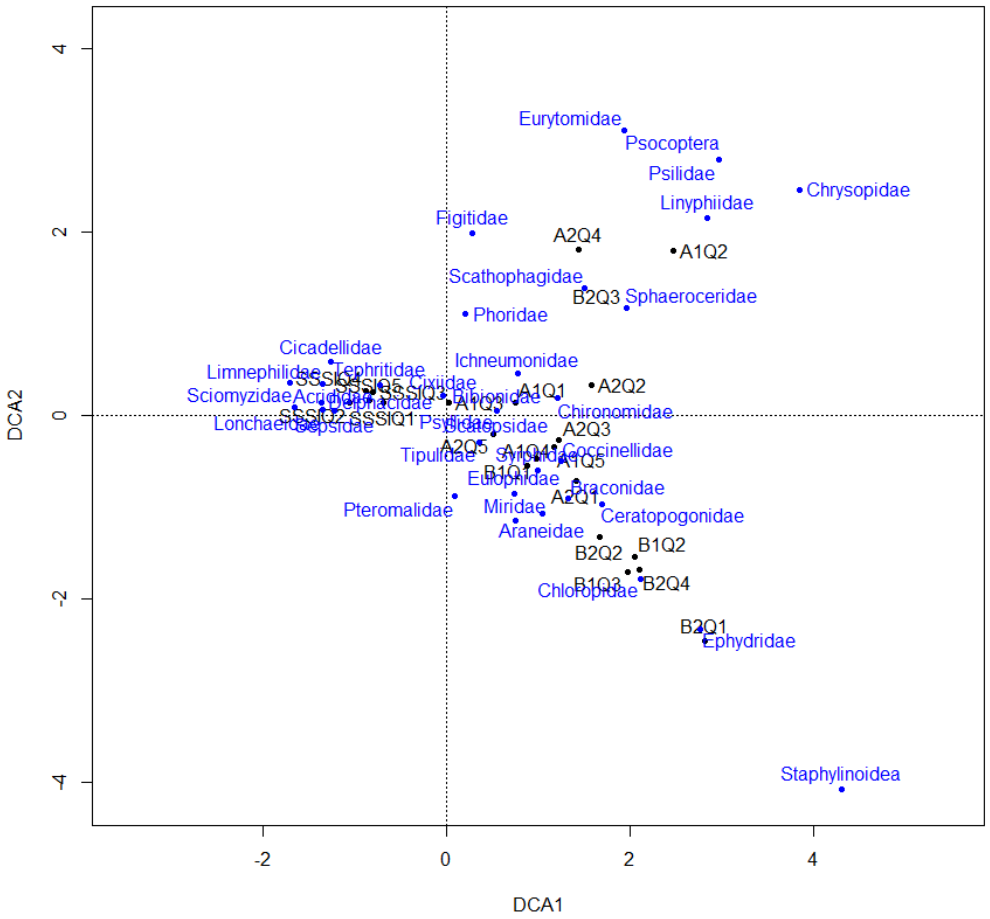


Figure 5.41 DCA of invertebrates caught in sweep nets in August 2016.

5.3.3.2.4 August 2017 (three years post-breach)

A2Q1 was clustered with SSSIQ2 in August 2017 (Figure 5.42), and they had similar scores on the first axis of the DCA ordination (Figure 5.43).

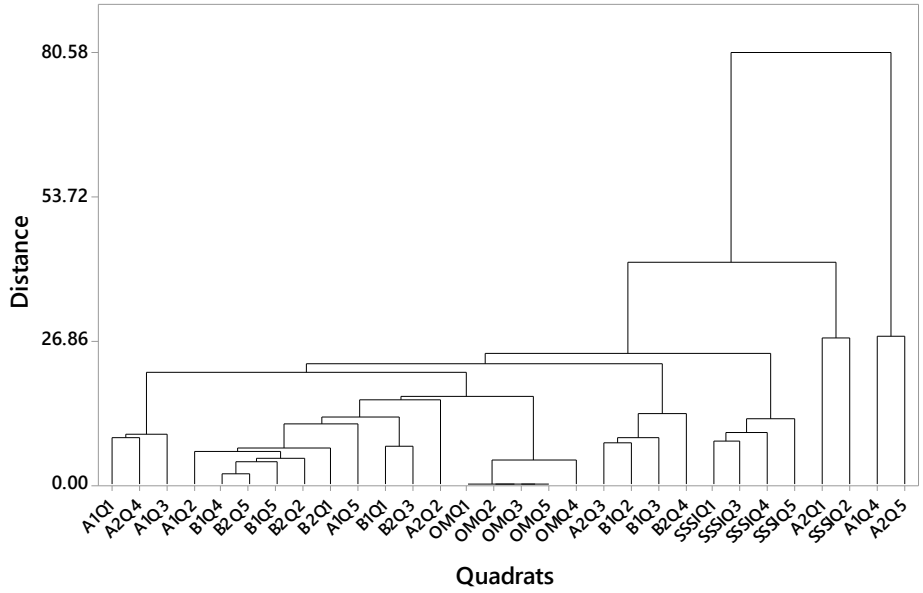


Figure 5.42 Clustering of study site quadrats using observations of number of individual invertebrates recorded at family level in sweep nets in August 2017. SSSI data are from August 2016. Average linkage and Euclidean distance were used to calculate clusters.

40% of the Site A quadrats and 60% of the Site B quadrats were close in distance to the SSSI by 2017 and had similar scores on the first axis of the ordination (eigenvalue = 0.5) (Table 5.19).

Table 5.19 Site scores on the DCA1 axis of sweep net August 2017 DCA analysis, showing similarities between scores in Steart Marsh quadrats and SSSI quadrats. All other quadrats had scores > 0.

DCA axis 1	Quadrat	Site scores
Eigenvalue = 0.5	SSSIQ1	-0.664
	SSSIQ2	-0.570
	SSSIQ3	-0.534
	SSSIQ4	-0.665
	SSSIQ5	-0.527
	A1Q2	-0.245
	A1Q4	-0.208
	A2Q1	-0.311
	A2Q5	-0.497
	B1Q1	-0.492
	B1Q4	-0.050
	B1Q5	-0.482
	B2Q2	-0.273
	B2Q3	-0.412
	B2Q5	-0.263

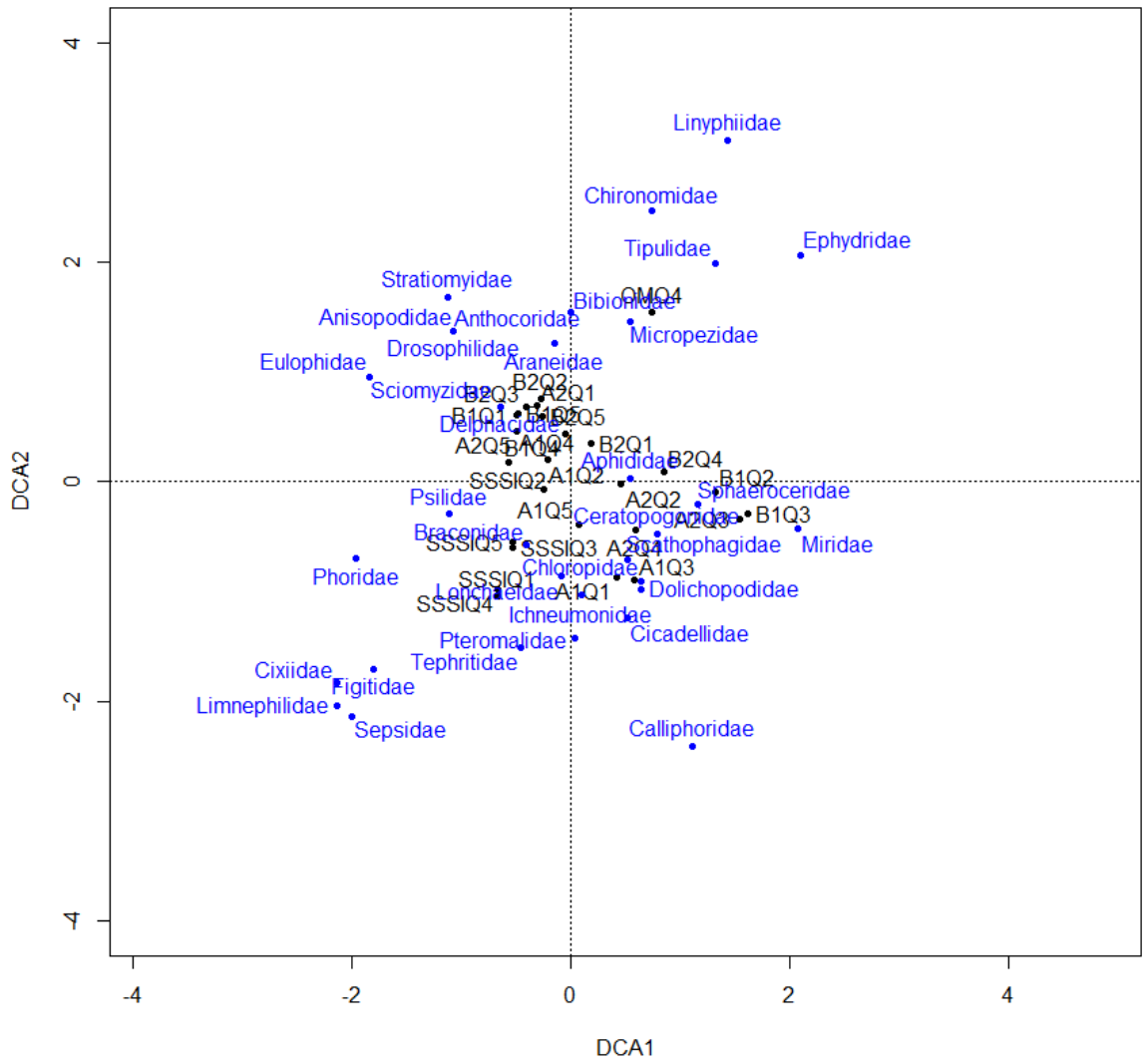


Figure 5.43 DCA of invertebrates caught in sweep nets in August 2017.

The Site B quadrats that had similar scores to the SSSI had high numbers of Delphacidae, which was like the SSSI, and the Site A quadrats that were similar to SSSI quadrat scores had high numbers of Delphacidae and Tephritidae flies.

A Procrustes plot shows how the quadrats changed in relation to plant-dwelling invertebrate assemblages in quadrats from August 2014-2017 (Figure 5.44). In general, Site A quadrats moved toward the SSSI quadrats, although one SSSI quadrat ordinated closer to some of the Site B quadrats, because of similarities in invertebrate families. OMQ4 was the only quadrat on OM where invertebrates were caught in 2017, but this moved in the opposite direction and away from the Steart Marsh quadrats and the SSSI.

A2 quadrats changed the most from the starting position in 2014 (Figure 5.45), but not all moved toward the SSSI quadrats. For example, there was high variation between years in A2Q3, but it is not similar to SSSI quadrats. Similar family groups and abundance may have been present between years where there is less variation between quadrats, but

these individuals are not likely to have been the same species in most instances, although some flying invertebrates may have travelled between systems, especially because the sites are in close proximity.

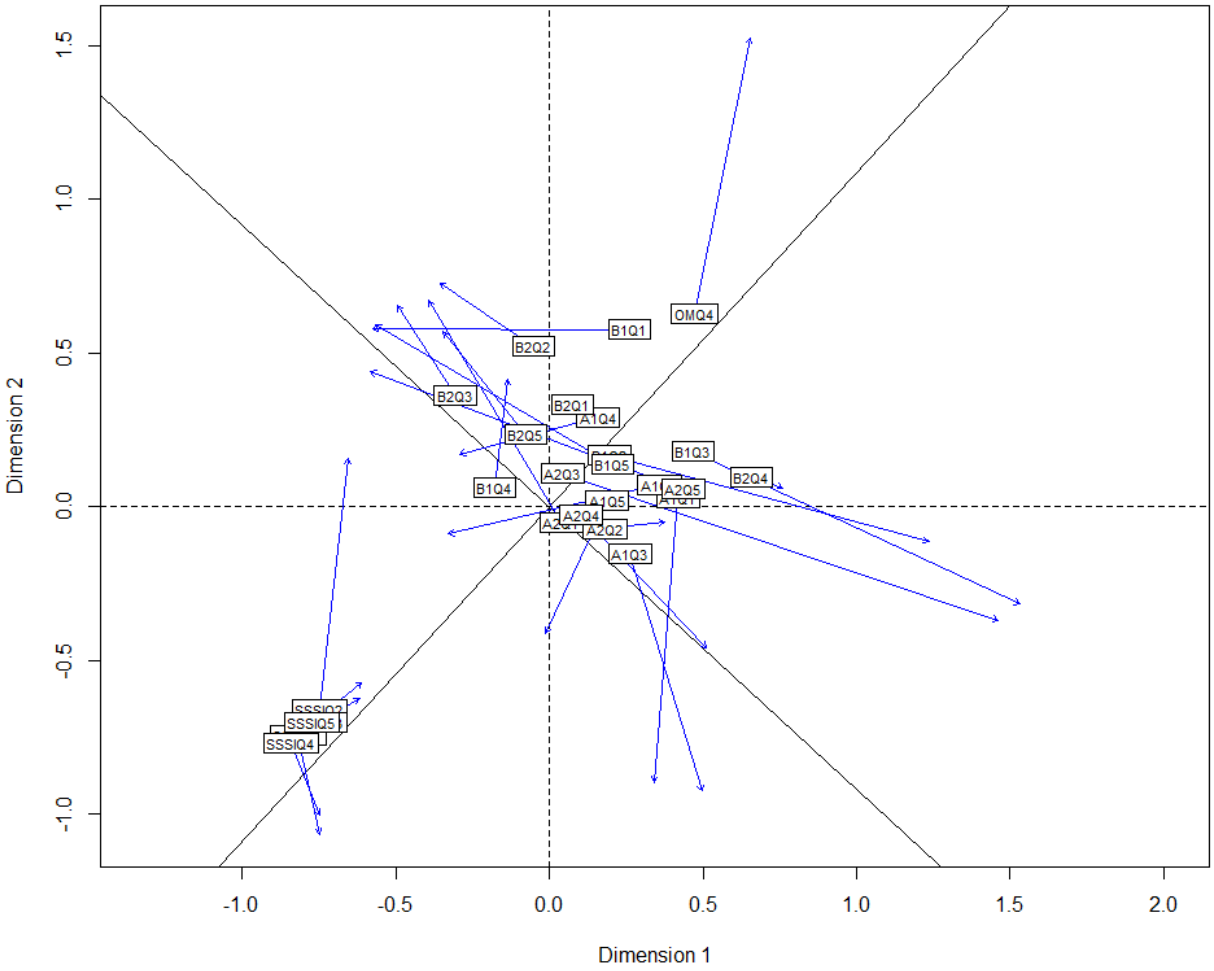


Figure 5.44 Procrustes plots showing a comparison of DCA ordinations of plant-dwelling invertebrates in August 2014 and 2017. Quadrat labels are the position of quadrats in 2014 and arrow heads are the position of quadrats in 2017. The SSSI data was the same in both ordinations (recorded in August 2016), but moved in relation to the other quadrats.

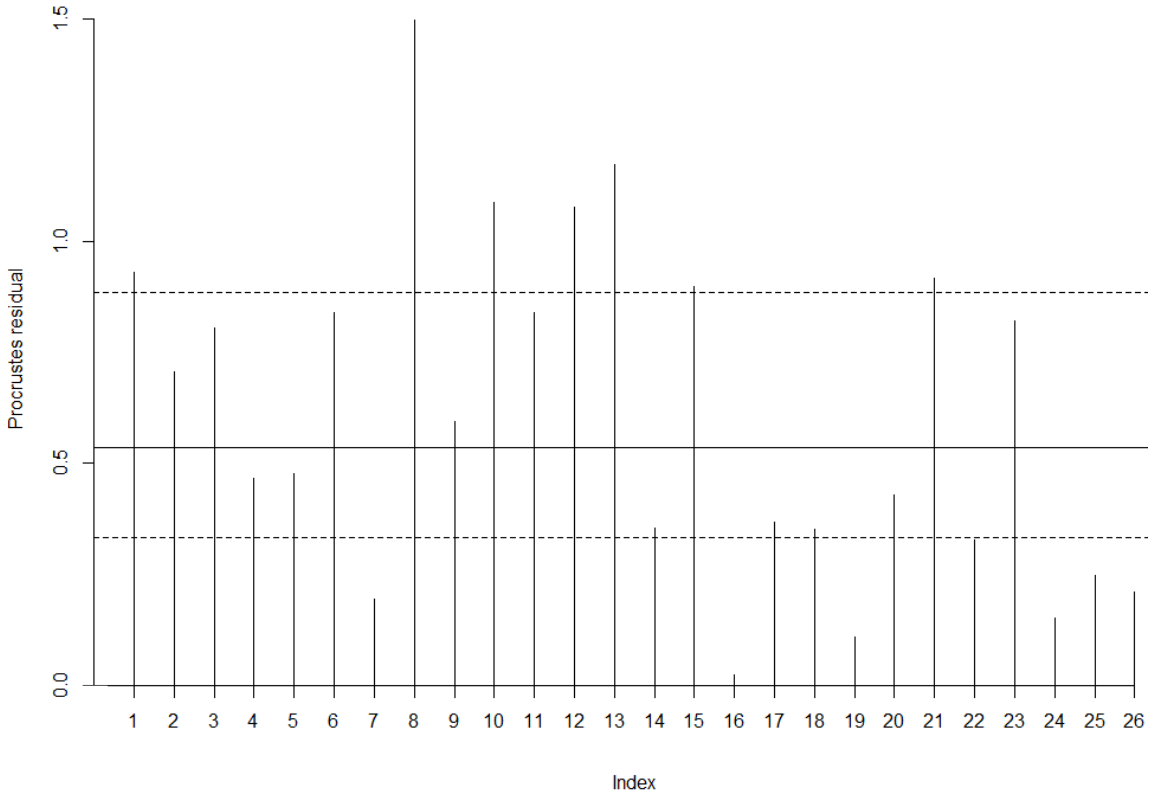


Figure 5.45 Procrustes error residuals for 2014 and 2017 plant-dwelling invertebrate ordinations. Index relates to quadrats: 1-5 = A1Q1-A1Q5; 6-10 = A2Q1-A2Q5; 11-15 = B1Q1-B1Q5; 16-20 = B2Q1-B2Q5; 21 = OMQ4; 22-26 = SSSIQ1-SSSIQ2. OMQ1, Q2, Q3 and Q5 were not included in the analysis because invertebrates were not caught in these quadrats. 25-75 % quantiles are between dotted lines and the solid line represents 50% quantiles of the residuals.

5.3.4 Invertebrate family richness

Total counts of invertebrate specimens caught in pitfall traps show that numbers on the SSSI were much higher than on other sites throughout the study, which is mainly due to the large numbers of amphipods that were caught in August 2016 on the mature saltmarsh (Figure 5.46a). Numbers of invertebrates dropped on Steart Marsh and Otterhampton after the breach and were especially low on the Site B plots in August 2015. Family richness of ground-dwelling invertebrates (caught in pitfall traps) and plant-dwelling invertebrates (caught in sweep nets) increased most notably on the Site A plots, and by August 2017 numbers and family groups on these plots were higher than they were in August 2014 (Figure 5.46a and Figure 5.47). Nevertheless, numbers of ground-dwelling invertebrates were still lower than the SSSI, chiefly due to the quantity of amphipod specimens that were caught.

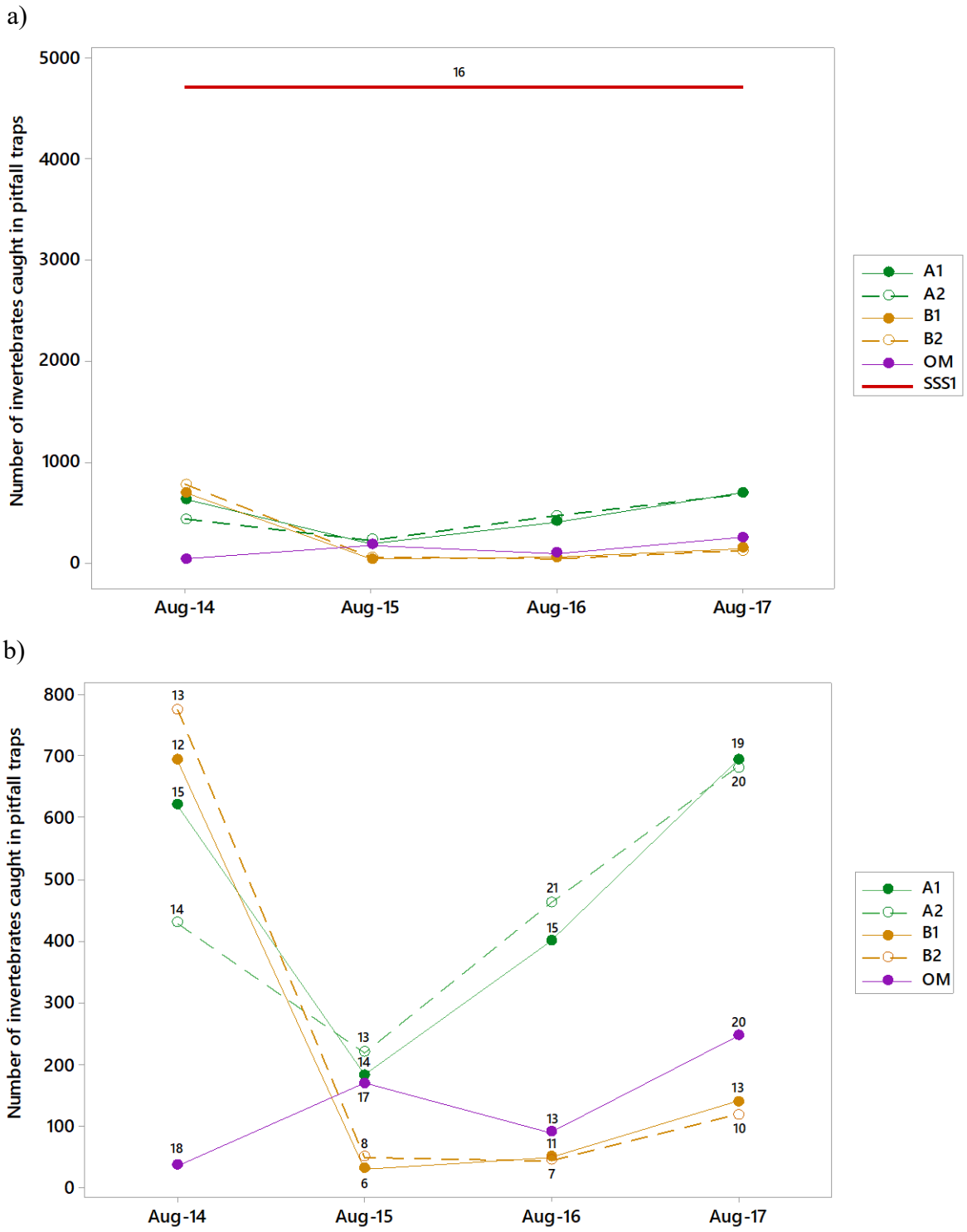


Figure 5.46 Number of invertebrates caught in pitfall traps on Steart Marsh study plots and OM over 9 day periods from August 2014-August 2017; a) including the SSSI (red line) as reference to the target state; b) same data with the SSSI removed to aid clarity. Numbers = quantity of family groups. OM family group numbers include one at order level (Pulmonata) on all sites in August 2014 and August 2015 on OM.

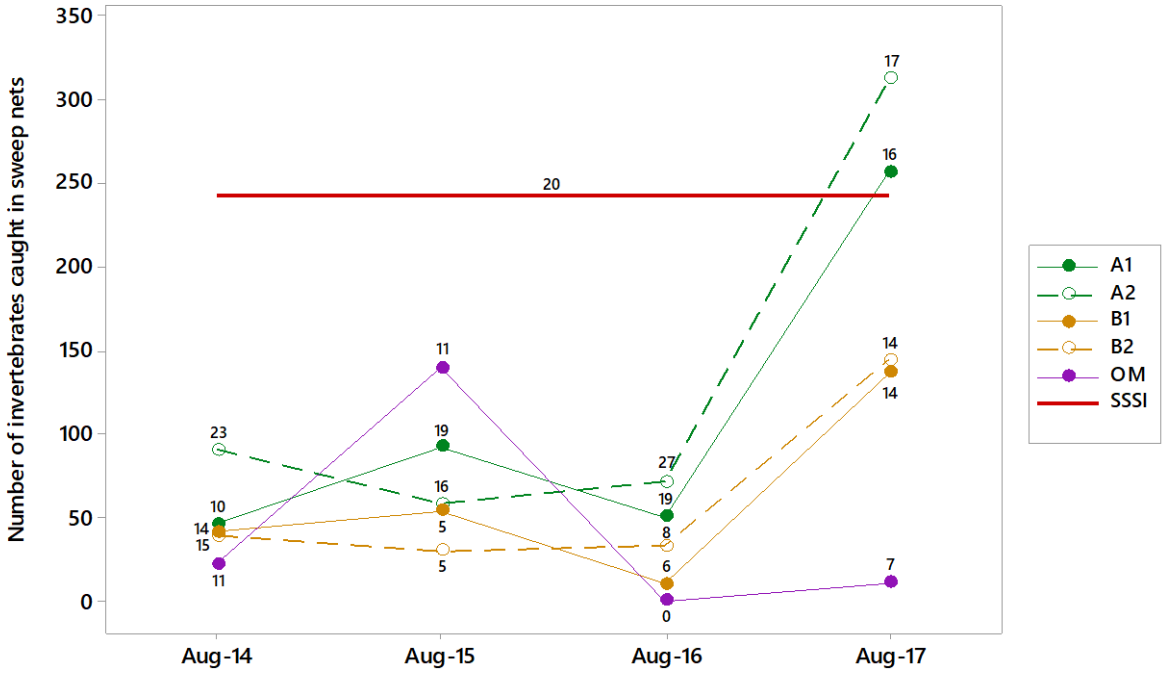


Figure 5.47 Number of invertebrates caught in sweep nets on Steart Marsh study plots and OM from August 2014-August 2017. Numbers = quantity of family groups. Red line represents the quantity of invertebrate specimens caught in sweep nets on the SSSI saltmarsh in August 2016.

5.3.5 Selected species

Data suggests that Talitridae were the most numerous family group found in pitfall traps on Steart Marsh plots (apart from B2) by 2017, and they were the most prevalent caught on the SSSI. In sweep net samples, Delphacidae was the most common family caught by August 2017 on Steart Marsh, and Tephritidae was especially common on the Site A plots. Talitridae, Tephritidae and Delphacidae were identified to species level due to their prevalence on these plots and the SSSI target state. Due to the amount of Talitridae specimens caught, fifty specimens were identified from each tube on the SSSI and compared to those found on Steart Marsh. It was determined that they were all of the same species, *Orchestia gammarellus,* which proliferated on the Site A plots in particular by the end of the study (Figure 5.48).

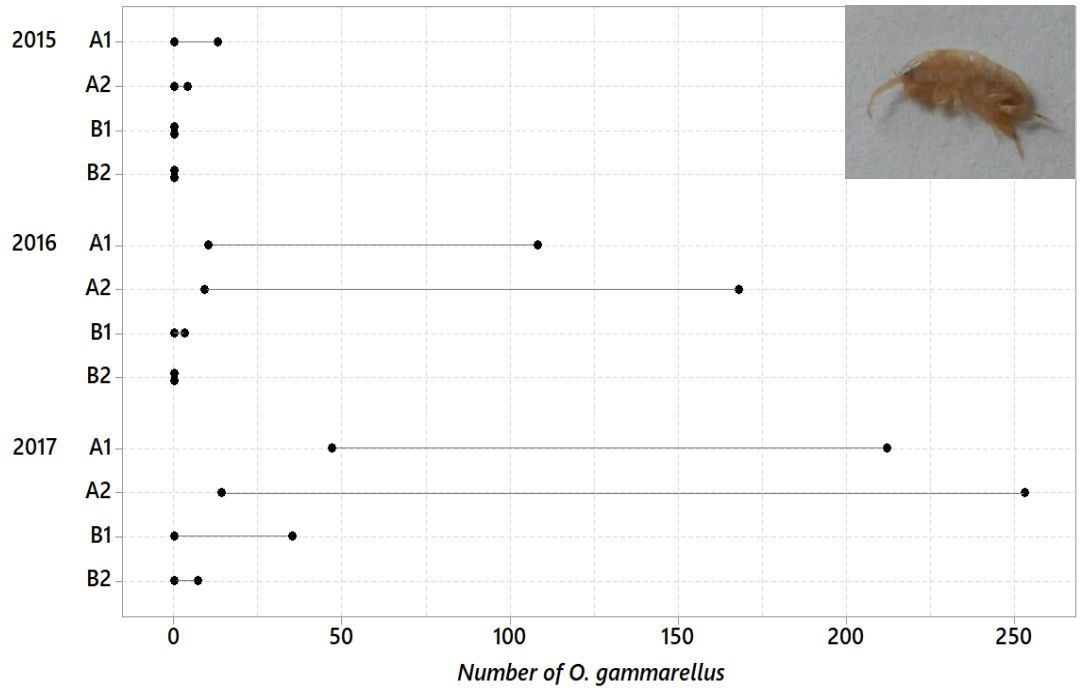


Figure 5.48 Number of *O. gammarellus* caught in pitfall traps on all Steart Marsh plots from August 2015 - August 2017, showing the spread of data between quadrats on each site (n = 5). Inset: *O. gammarellus* specimen caught on Steart Marsh (photograph taken by A. George, 2016).

The Tephritidae species found predominantly on the SSSI and the Site A plots was identified as *C. plantaginis*, which is associated with *Aster tripolium*.

To test if there was a correlation between *C. plantaginis* (Tephritidae) and its host plant *A. tripolium* over the course of the study, data were first tested for normality using an Anderson-Darling test. Data were found to be non-normal, so a Spearman Rank correlation was conducted. The test showed that there was a significant positive correlation between *A. tripolium* % cover and *C. plantaginis* abundance ($rs_{(78)}$ = 0.83, p <0.001) on all sites, including the SSSI, from August 2015 – August 2017 (SSSI was August 2016 data) (Figure 5.49). To determine if there was a correlation between the species on the newly created sites, SSSI data were removed from the dataset. It was found that there was a significant correlation between species and host on Steart Marsh and Otterhampton Marsh ($rs_{(73)}$ = 0.80, p<0.001).

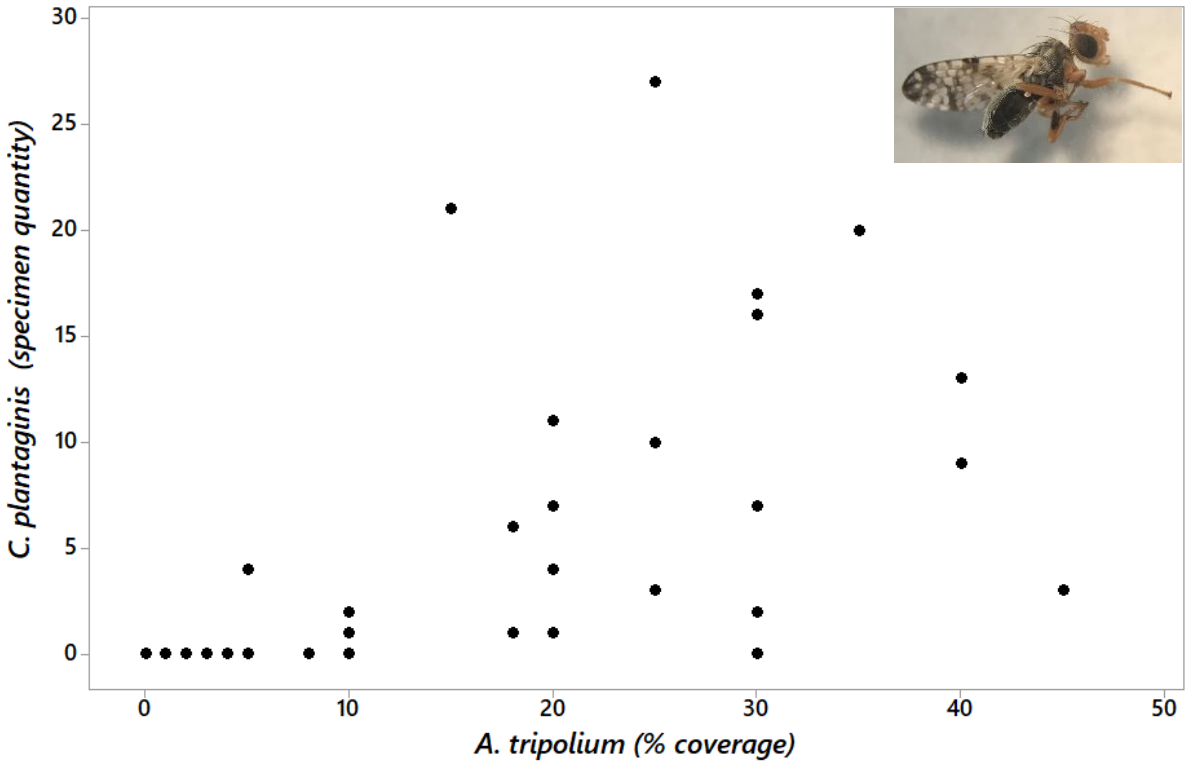


Figure 5.49 Relationship between *A. tripolium* and *C. plantaginis* caught in sweep nets on all study plots from August 2015 - August 2017 showing a positive correlation (n = 80; rs (78) = 0.83, p <0.001). SSSI data from August 2016 is included. Inset: *C. plantaginis* specimen caught on Steart Marsh.

Post-breach both *A. tripolium* and *C. plantaginis* were more abundant on the Site A plots than on all other plots apart from the SSSI. To test if there were differences in the coverage of *A. tripolium* and the abundance of *C. plantaginis* on the Site A plots by year, Kruskal-Wallis tests were conducted on both A plots combined (A1 and A2). This test was chosen because data were found to be non-normal. There was a significant difference between *A. tripolium* and year (Kruskal-Wallis, H(2) = 16.24, p < 0.001), and Dunn post hoc tests confirmed that there was a difference between *A. tripolium* cover on Site A plots in August 2015 (*Mdn* = 2.5) and August 2016 (*Mdn* = 20.0), p = 0.002, and between August 2015 and August 2017, p = 0.001. These differences were significant after a Holm correction. There was no difference in coverage between August 2016 (*Mdn* = 20) and August 2017 (*Mdn* = 20), p = 0.799 (Figure 5.50a).

There was also a significant difference between the number of *C. plantaginis* on the Site A plots and year (Kruskal-Wallis, H (1) = 6.89, p = 0.009). August 2015 data could not be tested because there were zero specimens, so this was removed from the analysis. Numbers were significantly higher in August 2017 (*Mdn* = 6.5) than in August 2016 (*Mdn* = 1.0) (Figure 5.50b).

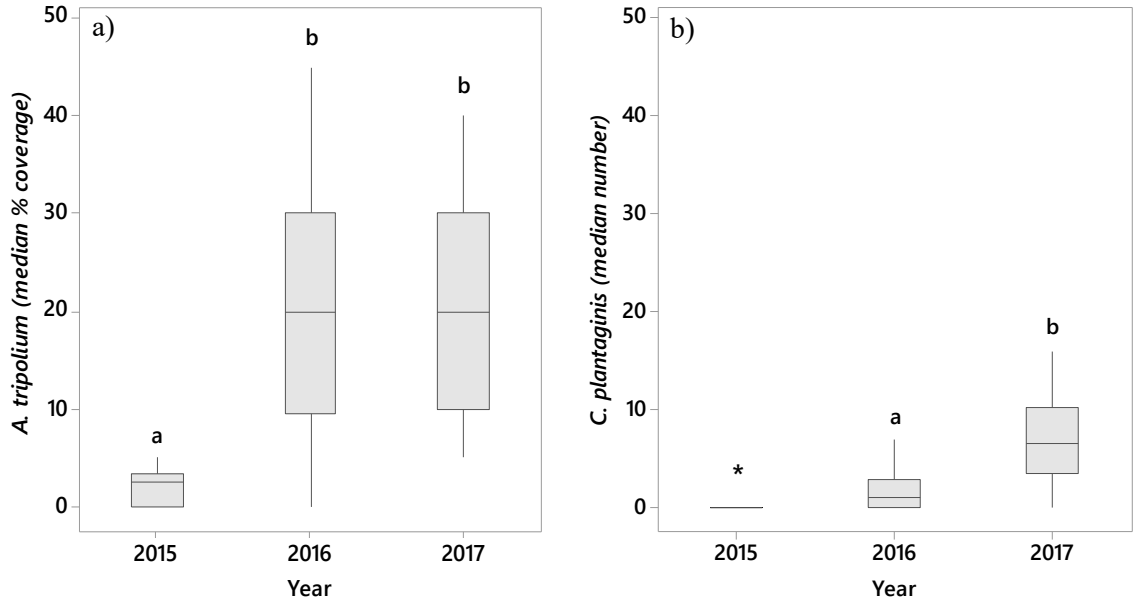


Figure 5.50 Median percentage coverage of *A. tripolium* and median number of *C. plantaginis* caught on Steart Marsh (Site A plots) post-breach; a) Median percentage coverage of *A. tripolium* on Site A plots (A1 and A2 combined) from August 2015 - August 2017 (n = 10 for each year); b) Median number of *C. plantaginis* on Site A plots (A1 and A2 combined) from August 2015 - August 2017 (n = 10 for each year). Medians that do not share a letter are significantly different ($p < 0.05$). * *C. plantaginis* were not present on the Site A plots in August 2015 so were removed from the analysis.

The Delphacidae specimens found in abundance on Steart Marsh is August 2017 was *Prokelisia marginata,* which has a close association with *Spartina anglica.*

There was a positive significant correlation between *S. anglica* and *P. marginata* across all sites and years ($rs_{(78)} = 0.60$, $p < 0.001$), and with SSSI data removed ($rs_{(73)} = 0.51$, $p < 0.001$) (Figure 5.51).

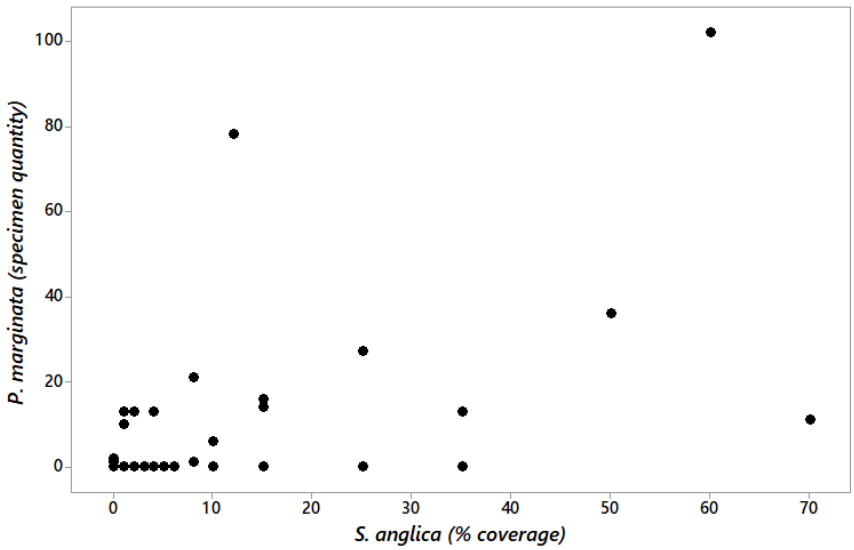


Figure 5.51 Correlation between *S. anglica* and *P. marginata* caught in sweep nets on all sites from August 2015 - August 2017 showing a positive correlation (n = 75; rs (73) = 0.51, p <0.001).

In general, the number of *P. marginata* specimens increased each year with *S. anglica*, but this was especially apparent in quadrats A2Q5, A1Q2 and A2Q1 (Figure 5.52). Both species were not present in all quadrats during the study, and they were absent on OM.

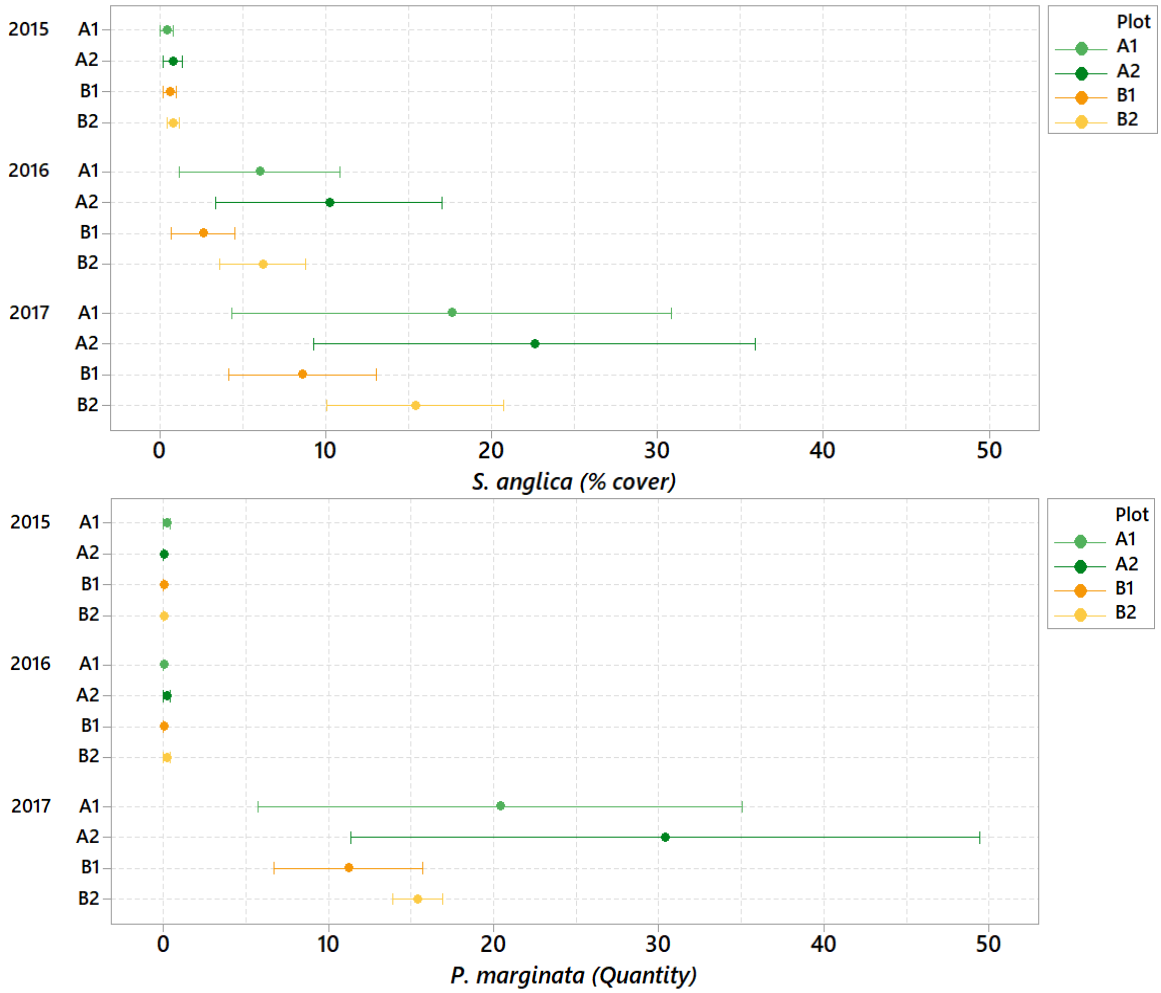


Figure 5.52 *S. anglica* versus *P. marginata* by year (August 2015-2017), caught in sweep nets in individual quadrats. Error bars represent standard error of the mean.

5.3.6 Soil-dwelling invertebrates

5.3.6.1 SSSI

On the SSSI 31 individual specimens were found in the top 20cm of soil. These individuals were distributed between three families, Enchytraeidae worms, Ellobiidae snails and Talitridae amphipods (Figure 5.53). Most of the specimens were in the top 10cm of soil (67.7%).

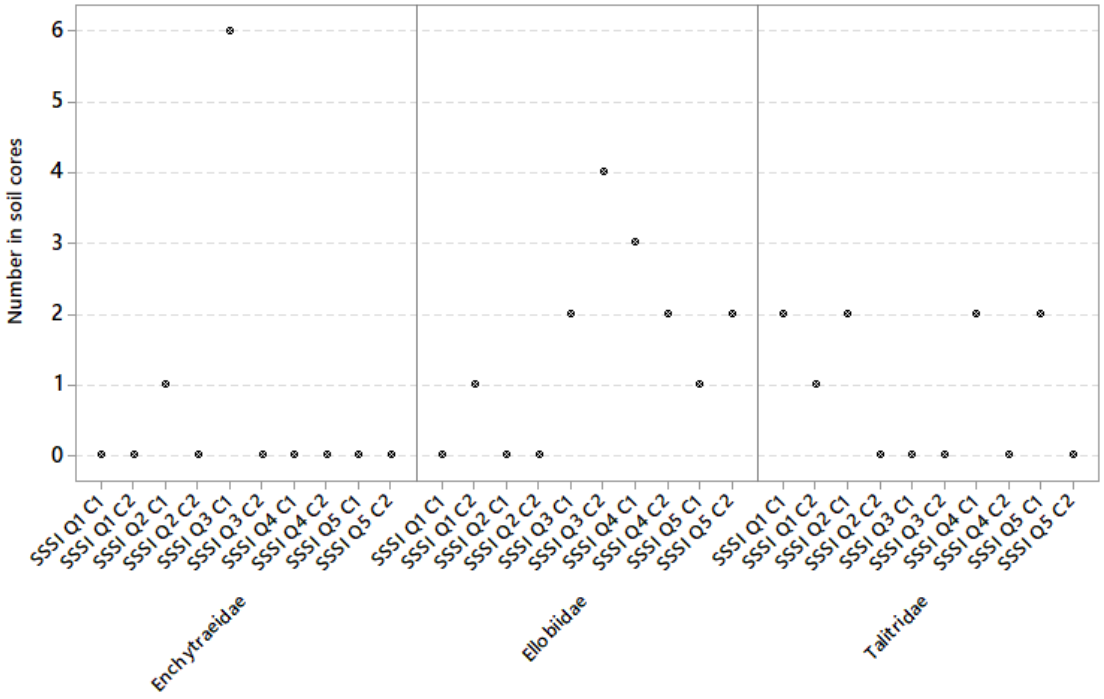


Figure 5.53 Number of invertebrates within family groups in SSSI cores. Recorded in 2016.

5.3.6.2 pre-breach

Soil cores for macro-invertebrates were collected each August, and it was surprising to find very few individuals within the August 2014 samples. From 50 samples, only 70 invertebrates were found. These were mainly found in the top 10cm of the soil and consisted primarily of annelid species. The lack of invertebrates found in the samples could be due to disturbance and potential compaction on the site, especially as more specimens were found on Site A plots (37) than Site B plots (18). 15 specimens were found on OM.

To assess the impacts of disturbance and compaction five core samples were taken from an adjacent pasture to assess the number of invertebrates as a comparison to the main site. From these soil samples 31 annelid specimen were found in the top 10cm of soil. This suggests that disturbance and compaction of the soil on the main site may indeed have been a factor in reducing the number of soil-dwelling invertebrate species.

5.3.6.3 Soil-dwelling invertebrates post-breach

Only one soil-dwelling invertebrate was found in soil cores in 2015. This was an enchytraeid worm in A1Q3C1. Numbers increased marginally in 2016 on the Site A plots, but soil-dwelling invertebrates were not found on the Site B plots or OM. On the A1 plot, one Talitridae amphipod and two Enchytraeidae worms were found in A1Q1C1, and one amphipod and one Enchytraeidae worm were found in A1Q3C1. On the A2 plot four

Enchytraeidae worms were found in A2Q4C1 and a Planorbidae shell was found in A2Q5C2.

Numbers of specimens increased in 2017 to 20 individuals on the A1 plot, and 12 on the A2 plot (Figure 5.54). In the A1 plot, 75% of invertebrate specimens caught were found in the top 10cm of soil (C1), and 100% of the 12 specimens caught on A2 were in C1 cores. There were no specimens caught in B2 soil, but one individual Enchytraeidae specimen was found on B1 in B1Q3C1. One Planorbidae shell was found on OM in OMQ2C2.

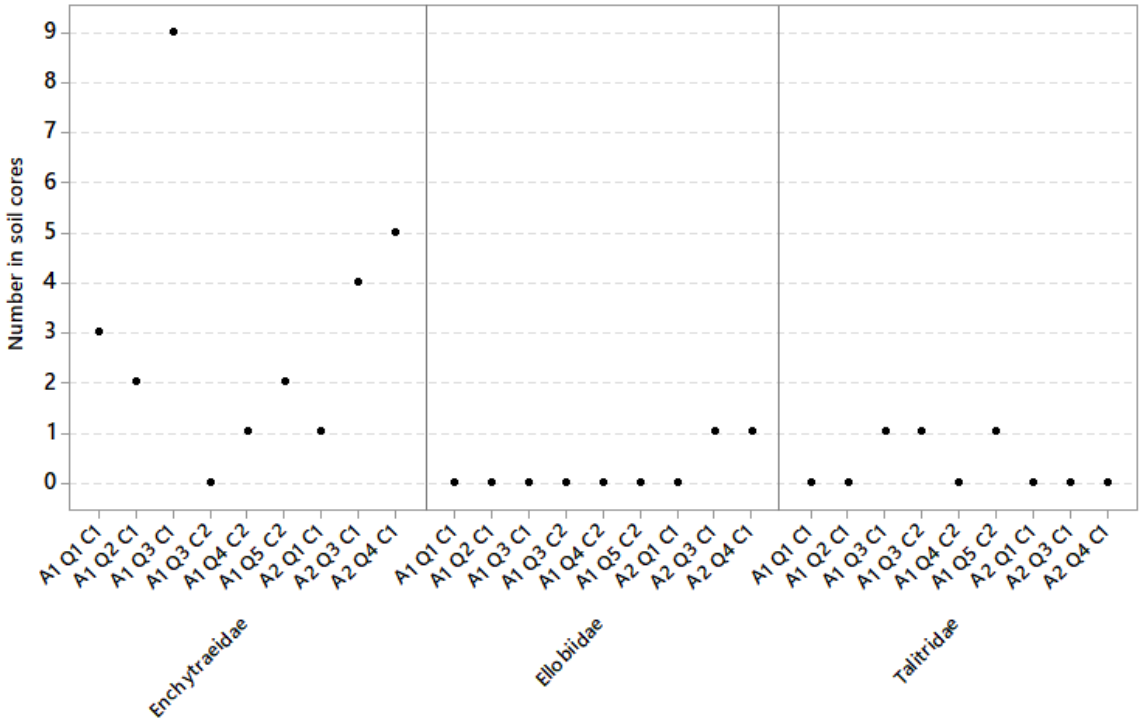


Figure 5.54 Number of invertebrates within family groups in A1 and A2 cores in August 2017. Cores that did not contain any invertebrates have not been included in the individual value plot.

5.4 Discussion

The aim of this chapter was to report and summarise ground-dwelling, flying and soil dwelling/benthic invertebrate species colonisation and composition on Steart Marsh in comparison to the managed OM site and the SSSI saltmarsh (target state), and determine trends of change in relation to the target state throughout the study period.

The key findings within this chapter are that several different family groups that were present on Steart Marsh and Otterhampton Marsh were absent shortly after the breach. However, similar family groups to those on the SSSI did colonise, although individual numbers were much lower. Site A was colonised more rapidly than all other sites and had the most similar invertebrate assemblages to the target state by the end of the study. *O.*

gammarellus specimens were numerous on the SSSI target state and they had colonised Site A plots at a faster rate than on all other plots. A dipteran species associated with sea aster that was found on the SSSI colonised Steart Marsh, especially on Site A plots where sea aster stands were more prevalent. It was absent from OM. An invasive plant hopper associated with *S. anglica* had spread across Steart Marsh in association with the colonisation of the plant. Invertebrate numbers were low on OM after the breach which is likely to be associated with anaerobic conditions that had affected vegetation growth.

Data gathered from each method of collection (pitfall traps, sweep nets and soil cores) have shown that the Site A plots on Steart Marsh are more comparable to the SSSI than all other sites after the three-year colonisation period post-breach. The reference site (OM) remained the most different in terms of invertebrate assemblages throughout the study. Higher quantities of individuals within characteristic saltmarsh fauna families are driving this change, including *O. gammarellus* (Talitridae), Tephritidae flies, and saltmarsh specialist plant hoppers (Delphacidae).

Amphipod numbers were relatively low compared to total counts on the SSSI plot in August 2017, but they were far more abundant on the Site A plots than they were on the Site B plots, and they were absent on OM. The high abundance of *O. gammarellus* on the SSSI was not unexpected, because it is a common saltmarsh species found on all coasts of North-West Europe (Hayward *et al.*, 2012). Its colonisation is important for saltmarsh functionality, as it is a food source for commercially important fish species such as sea bass (Cattrijsse and Hampel, 2006) and has been found in the digestive systems of bass on Steart during the study period (Stamp, 2017). It also plays a key role in nutrient cycling and aeration of the soil on the newly created intertidal area as it fills a similar ecological niche to earthworms in terrestrial systems (Schrama, 2012; Schrama et al., 2015). Research has shown that this species becomes the dominant invertebrate species on mature saltmarshes in the brown (detritus-based) food web (see Schrama, Berg and Olff, 2012), so it is encouraging to see such an important species colonising relatively rapidly, albeit only in large numbers on some of the sites.

Because of similarities in vegetation communities on the Site A plots and the SSSI by August 2017 (see Chapter 4), it is not surprising that the invertebrate species which colonised this area did so at a faster rate than the other sites on Steart Marsh and Otterhampton Marsh. However, numbers did start to increase on the Site B plots in August 2017, which suggests that amphipods will eventually be more abundant on these plots, too. The absence of amphipods on OM may be due to the lack of vegetation on the site, and

livestock grazing possibly had a negative influence because it has been shown to reduce numbers of the species due to the lack of shelter caused by trampling (Meyer *et al.*, 1995). However, expectations are not clear on OM because there is limited research on invertebrate assemblages on brackish sites, but it is clear that conditions are not comparable to the SSSI (see Chapter 3), and the distance of OM from mature saltmarshes may also be related to the absence of *O. gammarellus*. According to Petillon *et al.* (2014) amphipod numbers have been shown to be more than three times lower in MR sites than in mature saltmarshes, which is comparable to data gathered on the Steart peninsula during this study period.

The Tephritidae species found during this study, *C. plantaginis*, is associated with sea aster (*A. tripolium*), and its larvae are known to attack the plant (Brock, 2014). Adults of the species were found in relative abundance on the SSSI in August 2016 (98 individuals), and the species began to colonise the Site A plots in the same year. Numbers increased significantly on these plots by August 2017, but it was absent on all other sites. As sea aster was abundant on the Site A plots by 2017 it is reasonable to suggest that this is related to their colonisation, and there was a positive correlation between these species. *C. plantaginis* has been shown to reduce the seed dispersal capacity of sea aster (Castermans, Bossuyt and Bonte, 2008), which may not be beneficial to individual sea aster plants, but it could allow other plant species to colonise, thus enabling an increase in diversity of the local saltmarsh flora on Steart Marsh. It was noted during the study that *A. tripolium* plants were grazed by cattle, and the top parts of many of the plants were removed. According to Doody (2008) this can remove associated species, but data from this study show that *C. plantaginis* numbers did increase from August 2016 - August 2017 despite the site being grazed by English longhorn cattle from May 2017. Long-term monitoring of the species abundance will be necessary to determine if grazing has a negative impact on their numbers on Steart Marsh.

The invasive (first recorded in UK in 2008) plant-hopper (Delphacidae) species, *Prokelisia marginata,* that was caught in the study areas is monophagous on *Spartina* species, and its numbers were relatively high on the SSSI. The species was caught in pitfall traps and in high numbers in sweep nets on the Site A plots. In fact, higher numbers were caught on the Site A plots in August 2017 than on the SSSI the previous year. There was a correlation between this species and *S. anglica* on the study sites, and both species seem to be spreading quickly. *P. marginata* has spread rapidly along the British coast, starting in the south-east and it has recently been recorded on coasts in South Wales. If the spread of *S. anglica* is thought to be problematic, *P. marginata* may be beneficial, because it can

potentially limit the growth of *Spartina* species on the marsh, but could also indirectly affect saltmarsh stability, due to the role *Spartina* plays in preventing erosion. In a study by Grevstad *et al.* (2003) it was found that *P. marginata* decreased *Spartina* biomass by 50% compared to a control (Delphacidae-free condition), and the height of plants was also reduced in the presence of the species. The colonisation of *S. anglica* on the Steart Marsh MR sites could facilitate the spread of the species, because the plant is its primary host in Britain (Harkin, 2016).

Soldier flies (Stratiomyidae) were found abundantly on OM in August 2017, all of which were in the larval form. One individual flecked snout (*Nemotelus notatus*) specimen was found on OM in August 2016, and the larvae are from the same genus. *N. notatus* is commonly seen on estuarine marshes in the UK (Brock, 2014), but it was absent from all other sites during the study. Shore flies (Ephydridae) colonised the study sites in 2015, but numbers were much lower in the following years, although 50 specimens were caught in pitfall traps on OM in August 2017. There was a lack of vegetation on OM during this time, but some Ephydridae species feed on algae (Brock, 2014), which was present on OM at ground level. Barnard (2011) notes that members of this family often prefer to stay on the ground rather than fly, so their propensity to feed on algae on the soil may explain why they were more prevalent in pitfall traps on OM.

Lycosidae numbers were much lower on the sample plots by the end of the study than at the start. It was expected that there would be less diversity as the sites changed from disturbed agricultural land to an intertidal habitat, because flooding limits the foraging efficiency of hunting spiders (Döbel *et al.*, 1990). *Pardosa* spp. were the main constitute of the spider specimens caught on the SSSI, and the small numbers that were found on the sites by the end of the study were also mostly within this genus. However, the impact of cattle grazing on the site may also have influenced numbers, because grazing can reduce heterogeneity of vegetation (Pétillon *et al.*, 2007). The lack of Lycosidae on the study sites in August 2017 compared to the SSSI reference site differs from results gathered by Petillon and Garbutt (2008), who found that *Pardosa purbeckensis* were more prevalent on MR sites than reference marshes. Araneae species control numbers of other invertebrates and they do predate on Delphacidae, but Harkin (2016) suggests that the spread of the invasive *P. marginata* is unlikely to be limited by this pressure.

Few soil-dwelling invertebrates were found on the SSSI, and numbers were very low on the Steart Marsh and OM study plots in August 2015 and 2016. Numbers did rise on the Site A plots in 2017, and there were similarities with the SSSI, although the number of

specimens was lower. Only three families of invertebrates were found in the soil on the SSSI and the Site A plots: Enchytraeidae worms, Ellobiidae snails and Talitridae amphipods. Enchytraeidae are Oligochaeta worms, and they are often found in the stems of plants such as *Spartina* spp. (Healy and Walters, 1994). This could be why numbers were reasonably low in soil cores, but *Spartina* plants would have to be analysed to confirm this. There were many amphipods caught in pitfall traps, so it is not surprising that some were found in the soil. They would have burrowed into the soil from the surface, which helps to aerate the soil. Ellobiidae snails found on saltmarshes are omnivores and detritivores and evidence suggests that saltmarsh snails in this family are rarely found where there are bare patches of soil (Zajac *et al.*, 2017). Snails within this family were not found in cores of plot A1 in or prior to 2017, which had lower % bare ground than A2 sites, but one specimen was found in A2Q3 and in A2Q4, which had the lowest % bare ground of the A2 quadrats in 2017. Numbers of Elobiidae snails were higher on the SSSI that was fully vegetated.

It was expected that snails in the family Hydrobiidae would be found on the SSSI because they are a common occurrence on saltmarshes, but none were found in the core samples on the SSSI or Steart Marsh and OM during the study. Polychaete worms were also a notable absence in the soil in all plots. Using a sieving method Modley (2018) found specimens of Polychaete worms on Steart Marsh, but numbers were low. The absence (or low abundance) of Hydrobiidae snails and Polychaete worms on Steart may be of benefit because species within these taxa have had a negative impact on saltmarshes by curtailing plant succession and causing erosion (Andersen, 2001; Widdows and Brinsley, 2002; Paramor and Hughes, 2004; Widdows, Brinsley and Pope, 2009). However, they are also a food source for birds, so the apparent lack of soil-dwelling invertebrates is of concern, and bare ground that forms a salt pan is only beneficial to wading birds if invertebrates have colonised. The method adopted to record soil-dwelling invertebrates in this study has been used to good effect in studies of other wetland projects (T Shreeve personal communication, August 2014), and it worked successfully in the pasture adjacent to Steart Marsh. It is therefore unlikely that soil invertebrates were underestimated in cores. However, in future studies it may also be beneficial to carry out the more intensive sieving method adopted by Modley (2018).

To summarise, invertebrate colonisation on Steart Marsh follows a similar pattern to changes in soil characteristics and vegetation succession that was discussed in Chapter 3 and 4. The Site A plots have started to resemble the SSSI, and these changes toward the target state are most rapid on these plots. The invertebrates that are colonising Steart Marsh will

benefit natural processes that aid the productivity of the marsh, but there is also the likelihood that an invasive species will spread further on the marsh due to colonisation by *S. anglica.*

CHAPTER 6: COMMUNITY CHANGES AND PLANT FUNCTIONAL GROUPS

6.0 Introduction

Determining variation of community structures between different geographical sites is often an aim of ecological research (Guo, 2015), and it is important to assess this variation on an MR scheme to identify if there are differences between different starting positions. Although colonisation and community structure has been assessed on MR schemes (Wolters *et al.*, 2008; Hughes *et al.*, 2009; Mossman, Davy and Grant, 2012; Brooks *et al.*, 2015), few studies have incorporated soil, vegetation and invertebrates into analyses to assess how communities are structured on locations with different starting states in the early stages of development of a restoration scheme, and determine the abiotic characteristics that are driving change in comparison to a mature saltmarsh.

The aim of this chapter is to integrate data on soils, vegetation, and the invertebrate fauna to fully describe how Steart Marsh has changed over time since the site was breached with particular reference to the different starting states of each monitored area. In addition, the trajectories of the community changes are compared to understand what influences resemblance of study sites to the mature saltmarsh system (SSSI target state). Trait analysis is used within this chapter in relation to vegetation communities, because it helps to explain where individual plants can exist across the scheme (Guo, 2015), and total community change is intepreted in relation to key soil variables.

A multivariate approach was used to integrate the different data sets and address the multidimensional changes of the different plant and invertebrates components of the community in relation to soil, which is a key controlling factor. The results of this are interpreted to indicate the importance of different starting states on community change and thus management decisions within saltmarsh restoration schemes. Based on data from the previous chapters it is expected that a proportion of the Site A quadrats will be the most similar to the SSSI in terms of plant and invertebrate composition, which will be driven by similar abiotic factors (soil chracteristics).

6.1 Method

6.1.1 Canonical correspondence analysis

Canonical Correspondence Analysis (CCA) was used to assess how variation in species composition is explained by environmental variables. This multivariate technique

was conducted separately on vegetation, ground-dwelling invertebrates, plant-dwelling invertebrates, and soil-dwelling invertebrates. The environmental variables chosen for analysis were the seven soil characteristics detailed in Chapter 3 (Soil moisture, pH, EC, SOC (LOI), NO_3-N, NH_4-N, and PO_4-N). Variables in C1 and C2 cores were analysed with species composition separately. Vegetation presence/absence (including bare ground) was used in these analyses because an arch effect occurred with % data. Vegetation data were log + 1 transformed, but the arch effect persisted so it was decided that presence/absence data would be more appropriate for CCA analysis. Compaction and land height were assessed against vegetation in a separate CCA analysis. Bare ground was included in all vegetation CCA analysis because it was likely to represent a key difference between the changing sampled sites and the the fully vegetated target state. Invertebrate family abundances were log+1 transformed before running the analysis to minimise biases associated with the effect of rare or common family groups. Soil-dwelling invertebrate quantities were not transformed because species abundance was low.

6.1.2 RLQ analysis

Rates of change of vegetation communities and similarity of flora to the reference SSSI state was analysed using RLQ analysis, where R is a table of environmental variables, L is a species abundance or coverage table and Q is a species-trait table. The RLQ analysis was carried out on August 2017 C1 and C2 cores to gain an understanding of the functional groups at the end of the study. Trait data consisted of: clonality, maximum height of plants, seed weight, longevity, photosynthetic pathway, number of cotyledons, life form, and salt and moisture tolerance (based on Ellenberg indicators) (Table 6.1).

Table 6.1 Plant traits used in RLQ analysis with definitions. Information modified from Allaby (2004).

Plant trait	Definition
Clonality	
Creeping	Plants that spread by producing stolons
little spread	Plants that do not spread by stolons
Tussock	Plants that grow in clumps
Height (max)	Maximum height a plant can grow (cm)
Seed weight	Maximum weight of specific plant seeds (mg)
Longevity	
Annual	Completes lifecycle within one growing season
Perennial	A plant that lives for more than two seasons
Photosynthetic pathway	
C3	Common pathway of carbon fixation in plants in temperate regions
C4	Common pathway of carbon fixation in plants in tropical regions
CAM	(Crassulacean Acid Metabolism) C fixation in drought resistant plants
Cotyledon	
Monocot	One cotyledon (first leaf)
Dicot	Two cotyledons (first leaves)
Life form	
Chamaephyte	Buds or shoots are very close to the ground
Hemicryptophyte	Perennating buds are at ground level
Nanophanerophyte	Woody plants that have buds at a certain height above ground
Therophyte	Plant that completes its lifecycle during favourable conditions
Salt tolerance	How salt tolerant a plant is (based on Ellenberg values)
Moisture tolerance	Tolerance of a plant to waterlogging (based on Ellenberg values)

The traits used in this analysis are all specifically relevant to saltmarsh plants and similar traits were used in another study to assess traits in saltmarsh plants (see Guo *et al.*, 2015). However, specific traits were also chosen based on availability of information for every plant in this study. Trait data (Table 6.2) were gathered from the on-line Ecoflora database of British plants created by Fitter and Peat (1994).

Table 6.2 Trait values used in RLQ analysis. Information modified from Fitter and Peat (1994). LS = Little spread, C = creeping, T = Tussock; An = Annual, Per = Perennial; Mono = Monocot; Nano = Nanophanerophyte, Thero = Therophyte, Hemi = Hemicryptophyte, Cham = Chamaephyte. Salt and moisture tolerance were based on Ellenberg indicators.

Species	Clonality	Max Height (cm)	Longevity	Path	Seed weight (mg)	Cotyledon	Life from	Salt tolerance (Ellenberg)	Moisture tolerance (Ellenberg)
A.portulacoides	LS	80	An	C3	2.64	Dicot	Nano	6	8
A.prostrata	LS	100	An	C3	4	Dicot	Thero	2	7
A.tripolium	LS	100	An	CAM	0.41	Dicot	Hemi	5	8
C.anglica	LS	40	An	C3	1.31	Dicot	Hemi	6	8
E.atherica	C	120	Per	C3	4.53	Mono	Hemi	4	6
H.secalinum	T	70	Per	C3	5.96	Mono	Hemi	1	6
J.effusus	T	150	Per	C3	0.02	Mono	Hemi	0	7
L.perenne	T	90	Per	C3	2	Mono	Hemi	0	5
P.maritima	C	80	Per	C3	0.7	Mono	Hemi	5	8
S.anglica	C	130	Per	C4	12.1	Mono	Hemi	7	9
S.europaea	LS	30	An	C3	0.43	Dicot	Thero	9	8
S.marina	LS	20	An	C3	0.08	Dicot	Thero	5	8
S.maritima	LS	30	An	C3	0.79	Dicot	Thero	7	8
S.media	LS	30	An	C3	0.15	Dicot	Cham	5	8

Hillsmith PCA was used for the trait table in RLQ analysis as this type of PCA can be used when traits are categorical.

6.1.3 Non-metric multidimensional scaling

Non-metric multidimensional scaling (NMDS) was used to assess similarities of quadrats using Euclidean distance. This method was chosen instead of Principal Coordinate Analysis (PCoA) because the arch effect is associated with PCoA. Flora, soil characteristics and invertebrates caught within pitfall traps, sweep nets and soil cores were used as variables. This ordination showed how the sites were grouped by the end of the study (August 2017) in relation to the reference site (OM) and the target state (SSSI). All variables were standardised before analysis to avoid giving weights to specific variables.

6.1.4 Statistical analysis

CCA was carried out on the R Vegan package (Oksanen *et al.*, 2017), and RLQ analysis was conducted on the ade4package (Dray and Dufour, 2007; Dray and Legendre, 2008). Analysis of both CCA and RLQ analysis was conducted on R Studio. Non-metric multidimensional scaling was conducted on Past, version 3.16 (Hammer, Harper and Ryan, 2001).

6.2 Results

6.2.1 Canonical correspondence analysis

6.2.1.1 C1 cores

The CCA model of August 2014 data (Figure 6.1) was significant (p = 0.001) and showed that 53% of the variation in vegetation presence/absence was explained by soil variables. Variation on the first and second axes was non-random (both p = 0.001), but no other axes significantly contributed to the ordination. Of the seven environmental variables in the model, soil moisture (p = 0.001), EC (p = 0.001), NH_4-N (p = 0.023) and PO_4-P (p = 0.049) were significant after 999 permutations. This model reflects the difference between the starting states, with the terrestrial sites being on the opposite side of the environmental gradient to the SSSI (CCA1). This gradient is primarily influenced by soil moisture and salinity. Sites and species are grouped by macronutrients on CCA2.

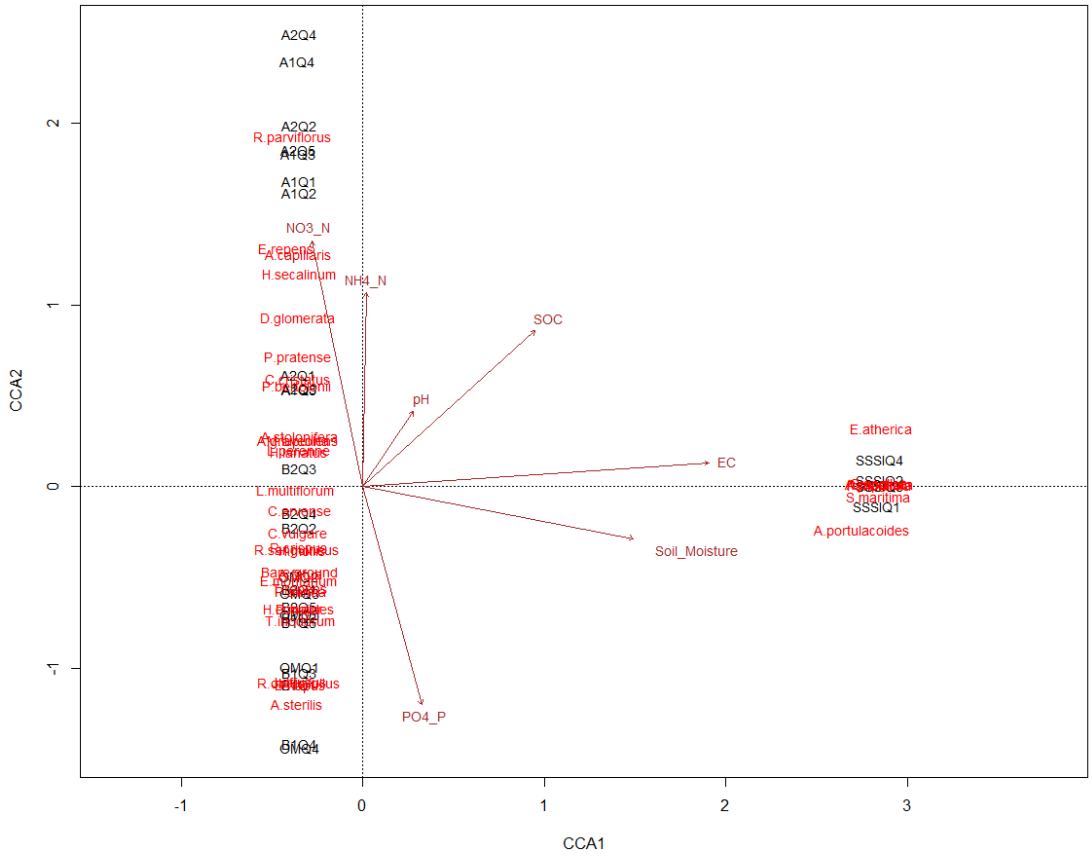


Figure 6.1 CCA ordination of plant species and C1 soil variables in August 2014.

The CCA model of August 2015 data (Figure 6.2) was significant (p = 0.001) and showed that 50% of the variation in vegetation presence/absence was explained by soil variables. Variation on the first axes was non-random (p = 0.001), no other axes significantly contributed to the ordination. Soil moisture (p = 0.001), EC (p = 0.001), and pH (p = 0.023)

were significant after 999 permutations. OM quadrat groupings were influenced by remnants of non-halophytic species and lower measurements of the chosen variables. Soil moisture, EC and pH were responsible for the grouping of quadrats in relation to the SSSI on the first axis.

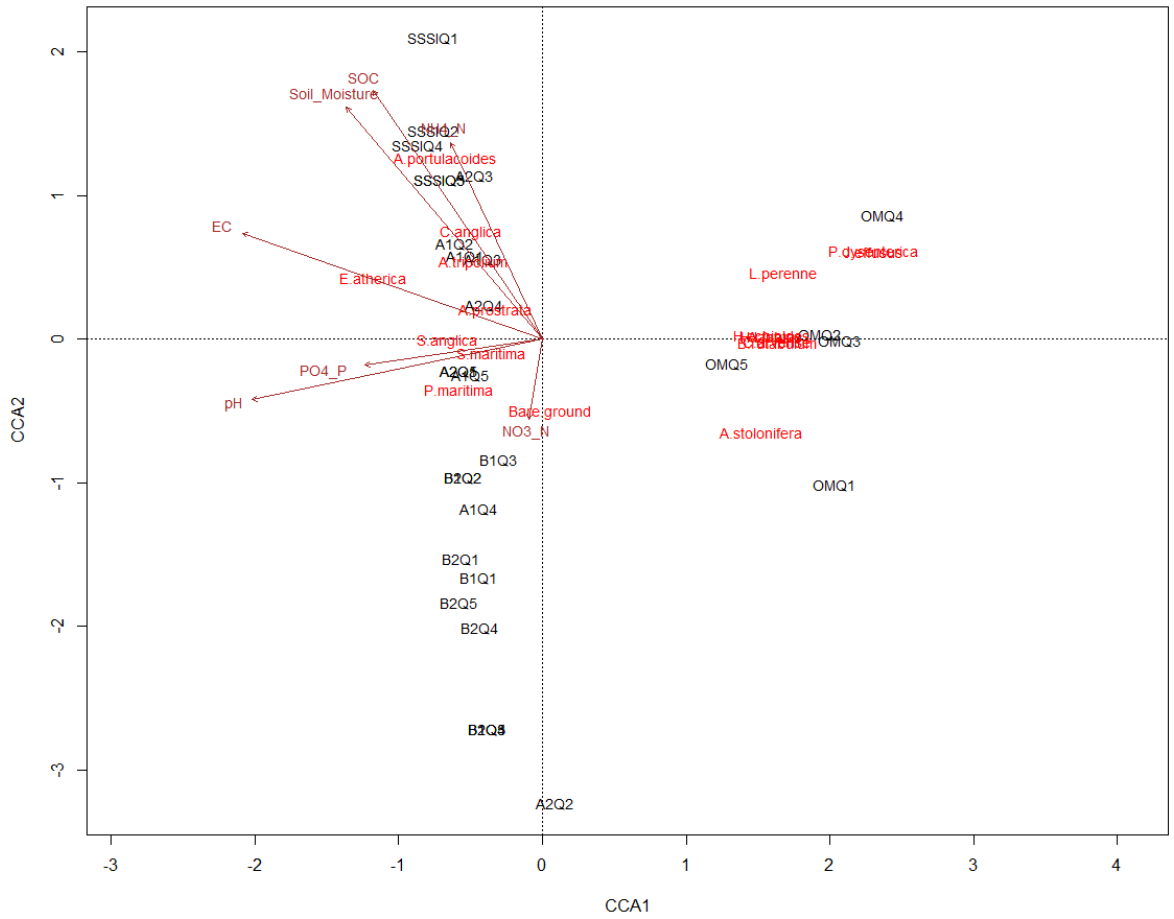


Figure 6.2 CCA ordination of plant species and C1 soil variables in August 2015.

The CCA model of August 2016 data (Figure 6.3) was significant (p = 0.001) and showed that 39% of the variation (less than previously) in vegetation presence/absence was explained by soil variables. Variation on the first and second axes was non-random (both p = 0.001), but no other axes significantly contributed to the ordination. Soil moisture (p = 0.001), EC (p = 0.015), and SOC (p = 0.018) were significant. SOC was having a major influence on the second axis, because levels were higher than on all other plots. Moisture and EC still had a major influence on the groupings on the first axis.

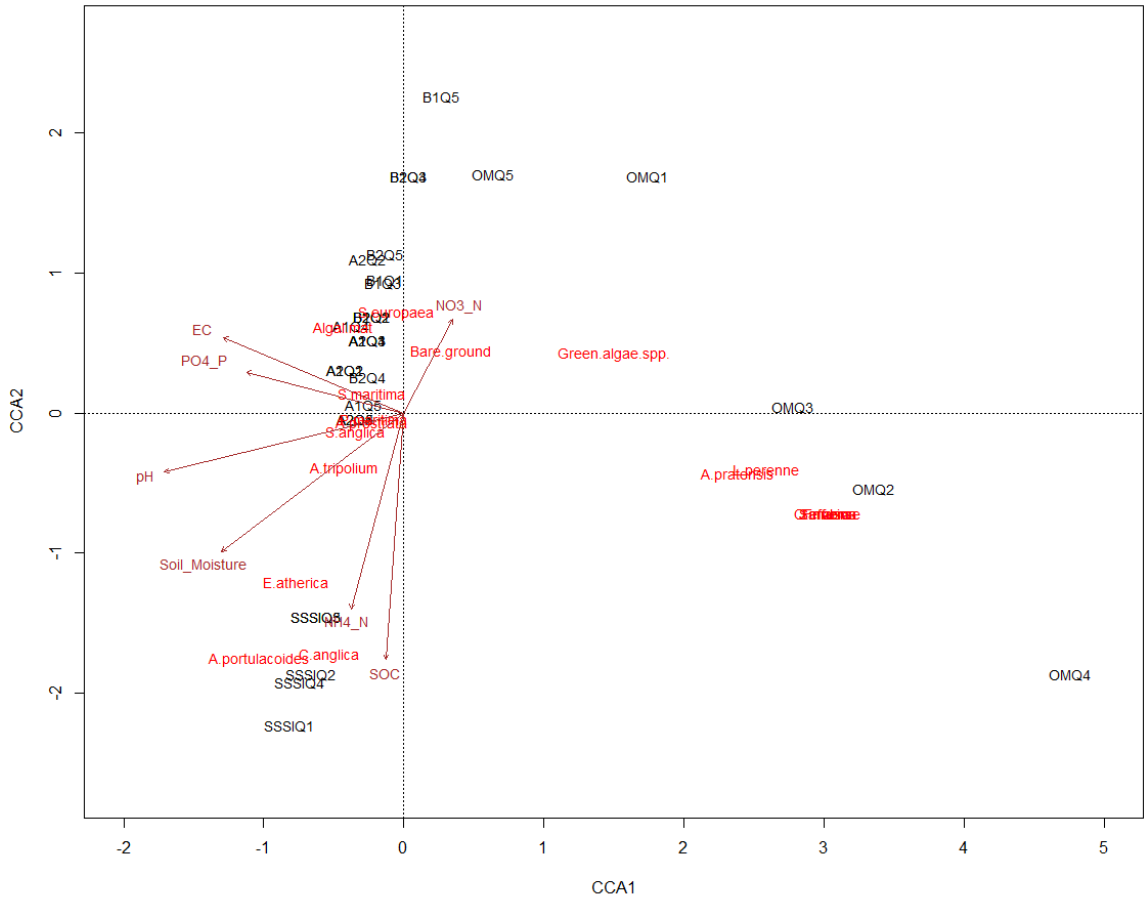


Figure 6.3 CCA ordination of plant species and C1 soil variables in August 2016.

The CCA model of August 2017 data (Figure 6.4) was significant (p = 0.001) and showed that 50% of the variation in vegetation presence/absence was explained by soil variables. Variation on the first and second axes was non-random (p = 0.001; p = 0.003), but no other axes significantly contributed to the ordination. Of the seven environmental variables in the model, soil moisture (p = 0.001), SOC (p = 0.042), NO_3-N (p = 0.023), PO_4-P (p = 0.037) and pH (p = 0.011) were significant after 999 permutations.

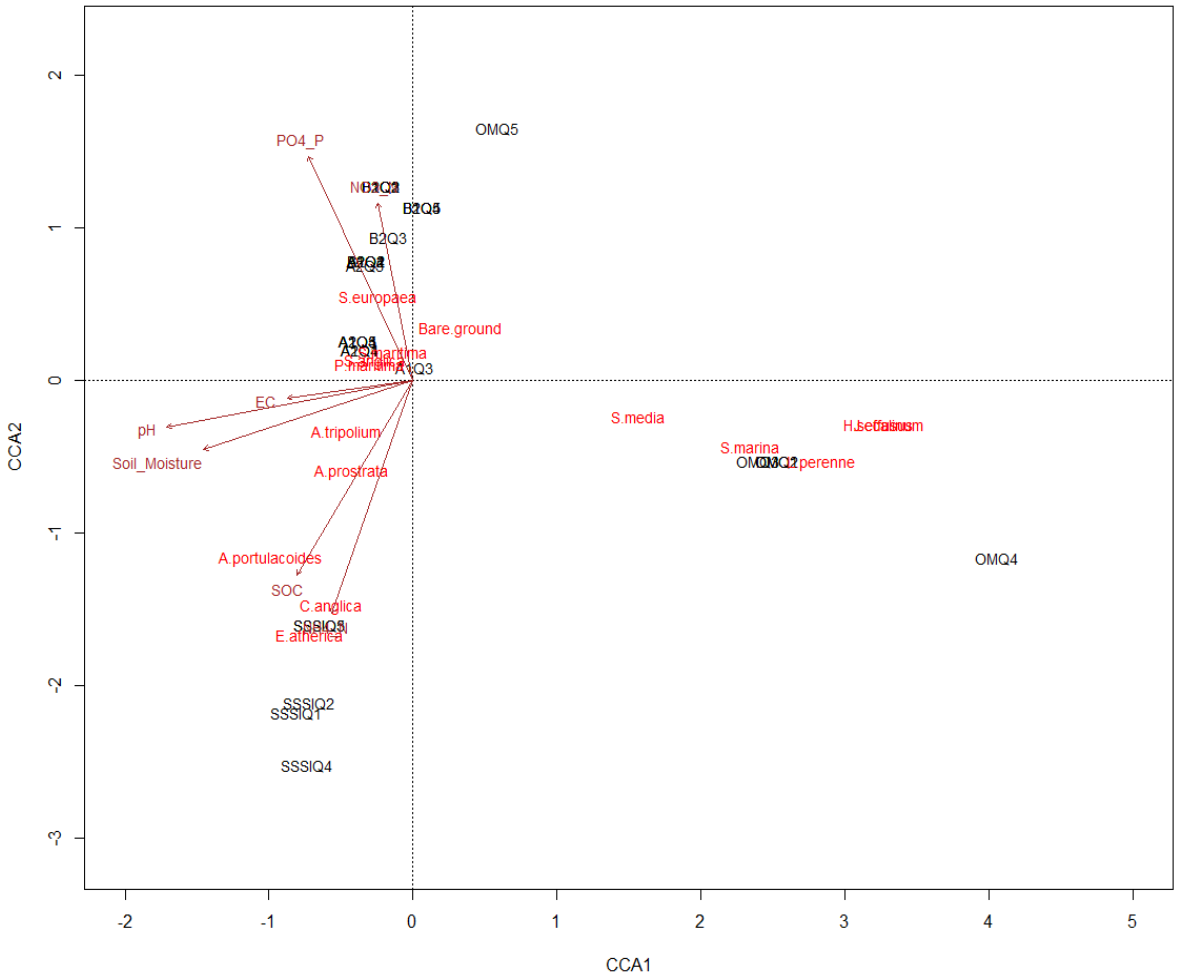


Figure 6.4 CCA ordination of plant species and C1 soil variables in August 2017.

6.2.1.2 C2 cores

The CCA model of August 2014 data (Figure 6.5) was significant ($p = 0.001$) and showed that 50% of the variation in vegetation presence/absence was explained by soil variables. Variation on the first and second axes was non-random ($p = 0.001$; $p = 0.030$), but no other axes significantly contributed to the ordination. Of the seven environmental variables in the model, soil moisture ($p = 0.001$) and EC ($p = 0.001$) were significant after 999 permutations. This followed a similar pattern to the August 2014 C1 data.

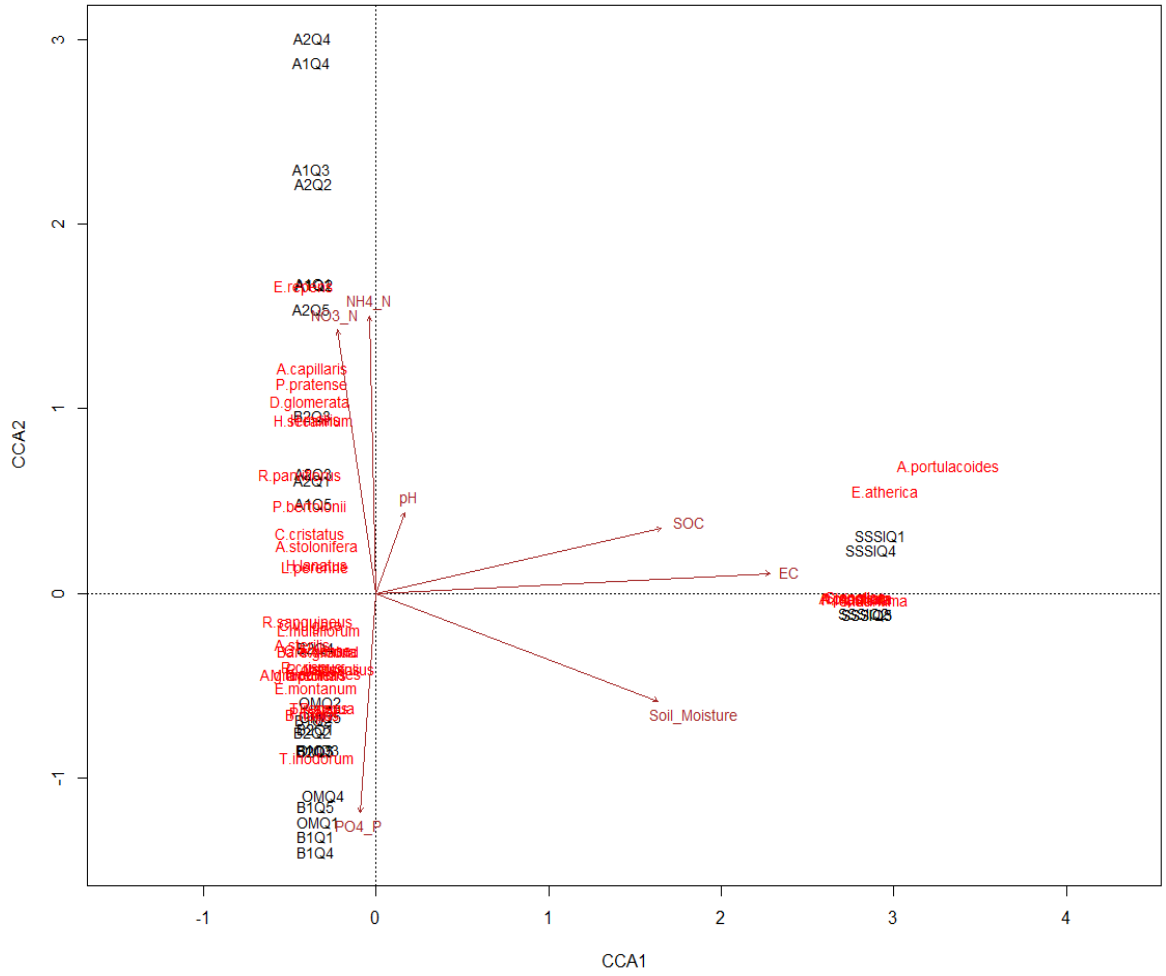


Figure 6.5 CCA ordination of plant species and C2 soil variables in August 2014.

The CCA model of August 2015 data (Figure 6.6) was significant (p = 0.001) and showed that 50% of the variation in vegetation presence/absence was explained by soil variables. Variation on the first axis was non-random (p = 0.001), but all other axes were not significantly different to random. Of the seven environmental variables in the model, soil moisture (p = 0.005), EC (p = 0.001), and SOC (p = 0.042) were significant after 999 permutations. The CCA ordination clearly shows the differences between Steart Marsh quadrats and OM quadrats only one-year after the breach.

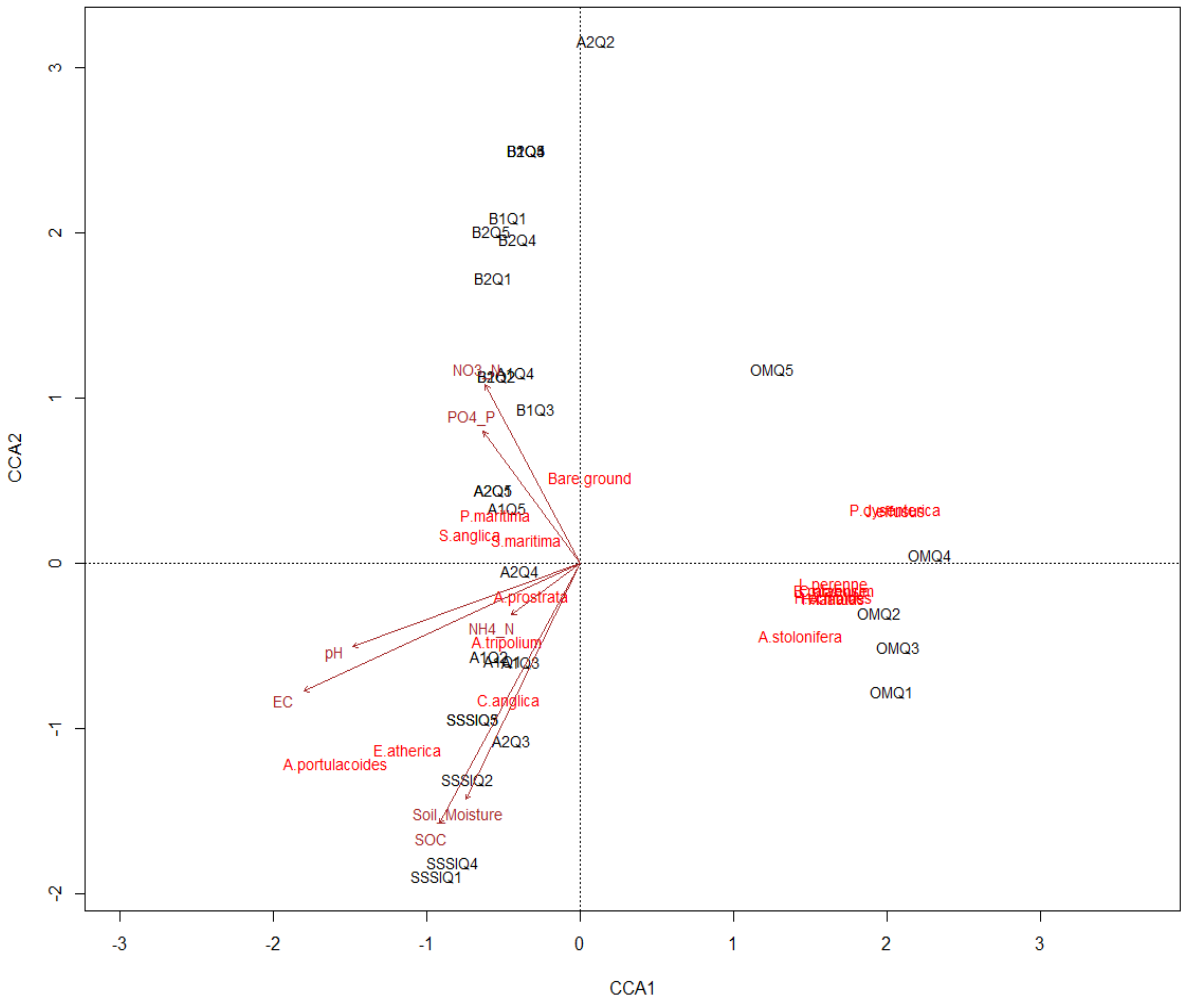


Figure 6.6 CCA ordination of plant species and C2 soil variables in August 2015.

The CCA model of August 2016 data (Figure 6.7) was significant ($p = 0.005$) and showed that 36% of the variation in vegetation presence/absence was explained by soil variables. Variation on the first axis was non-random ($p = 0.002$), but all other axes were not significantly different to random. Of the seven environmental variables in the model, soil moisture ($p = 0.003$) and EC ($p = 0.002$) were significant after 999 permutations. SOC was close to significance ($p = 0.055$). The Site A quadrats were marginally closer to the SSSI on the first axis compared to the Site B quadrats, which was driven by these two significant soil variables. Most OM quadrats were very different to other quadrats on the first axis.

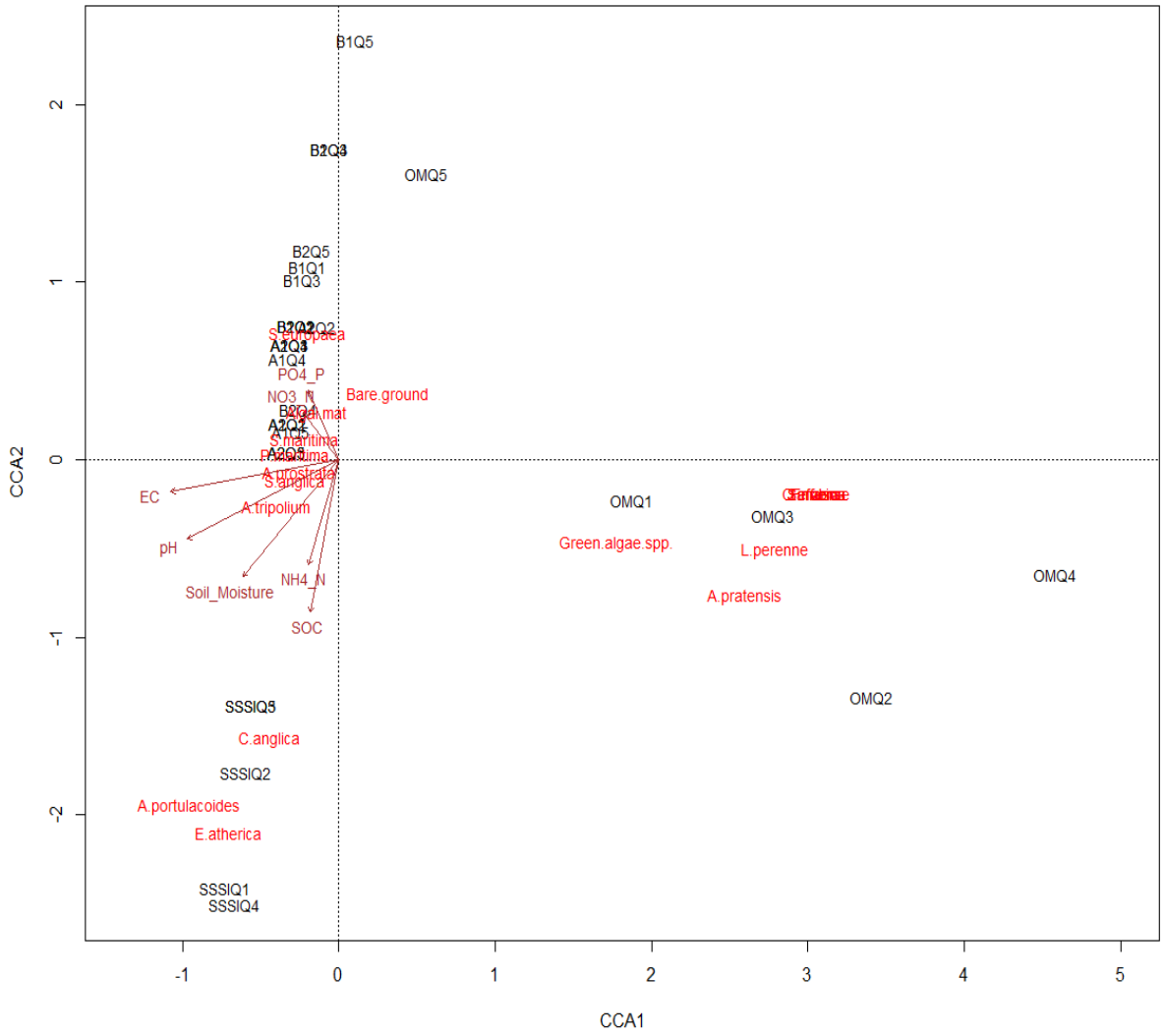


Figure 6.7 CCA ordination of plant species and C2 soil variables in August 2016.

The CCA model of August 2017 data (Figure 6.8) was significant (p = 0.001) and showed that 50% of the variation in vegetation presence/absence was explained by soil variables. Variation on the first and second axes was non-random (p = 0.001; p = 0.002), but no other axes significantly contributed to the ordination. Of the seven environmental variables in the model, soil moisture (p = 0.003) and EC (p = 0.001) were significant after 999 permutations. Site A quadrats were closer to the SSSI quadrats on the first and second axes, which were predominantly explained by both these soil variables.

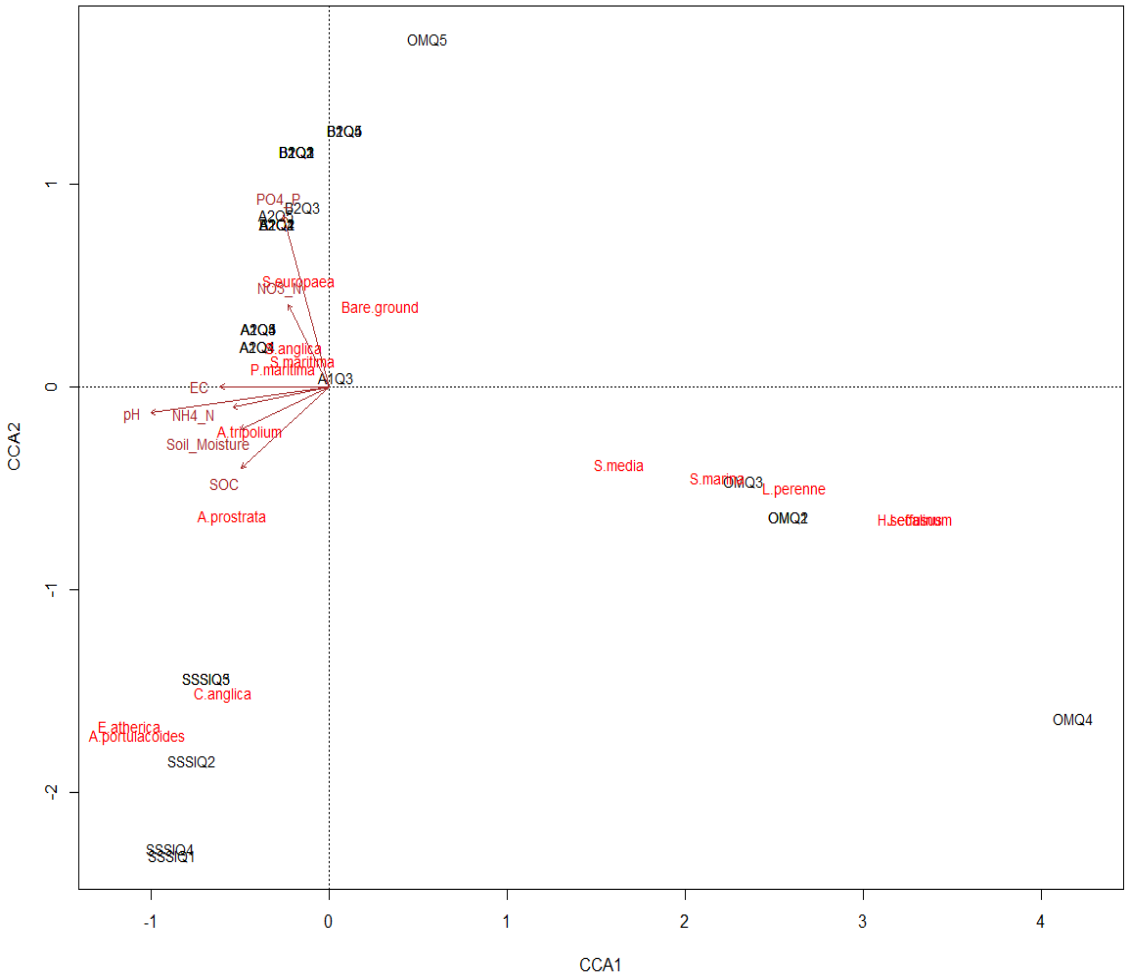


Figure 6.8 CCA ordination of plant species and C2 soil variables in August 2017.

6.3 RLQ analysis

6.3.1 C1 cores

The first two axis in the RLQ analysis were responsible for 97.1% of the variation, but the first axis was the most meaningful (eigenvalue > 1). All soil variables had negative correlations with the first axis, and most variables had negative correlations with the second axis, apart from NO_3-N and PO_4-P, which had a positive correlation > 0.6 (Table 6.3).

Table 6.3 C1 Soil variable correlations on RLQ axes. Figures in red font are negative correlations.

Soil variable	Axis1	Axis2
Soil moisture	-0.7845	-0.5758
pH	-0.8208	-0.4418
EC	-0.6351	-0.0957
SOC	-0.3795	-0.7989
NO3_N	-0.1800	0.6552
NH4_N	-0.2826	-0.8817
PO4_P	-0.4241	0.6705

The creeping clonality trait had a negative correlation with the first axis, but it was positively correlated with the second. Height, CAM and the hemicryptophyte and

nanophanerophytes life form traits were negatively correlated on both axes, whereas C3 and the chamaephyte and therophyte life form traits were positively correlated on both axes. The other traits that had positive relationships with the first axis were the annual and dicot traits, which were both negatively correlated on the second axis (Table 6.4).

Table 6.4 Plant trait correlations on RLQ axes related to C1 soil variables and plant presence absence data. Figures in red font are negative correlations.

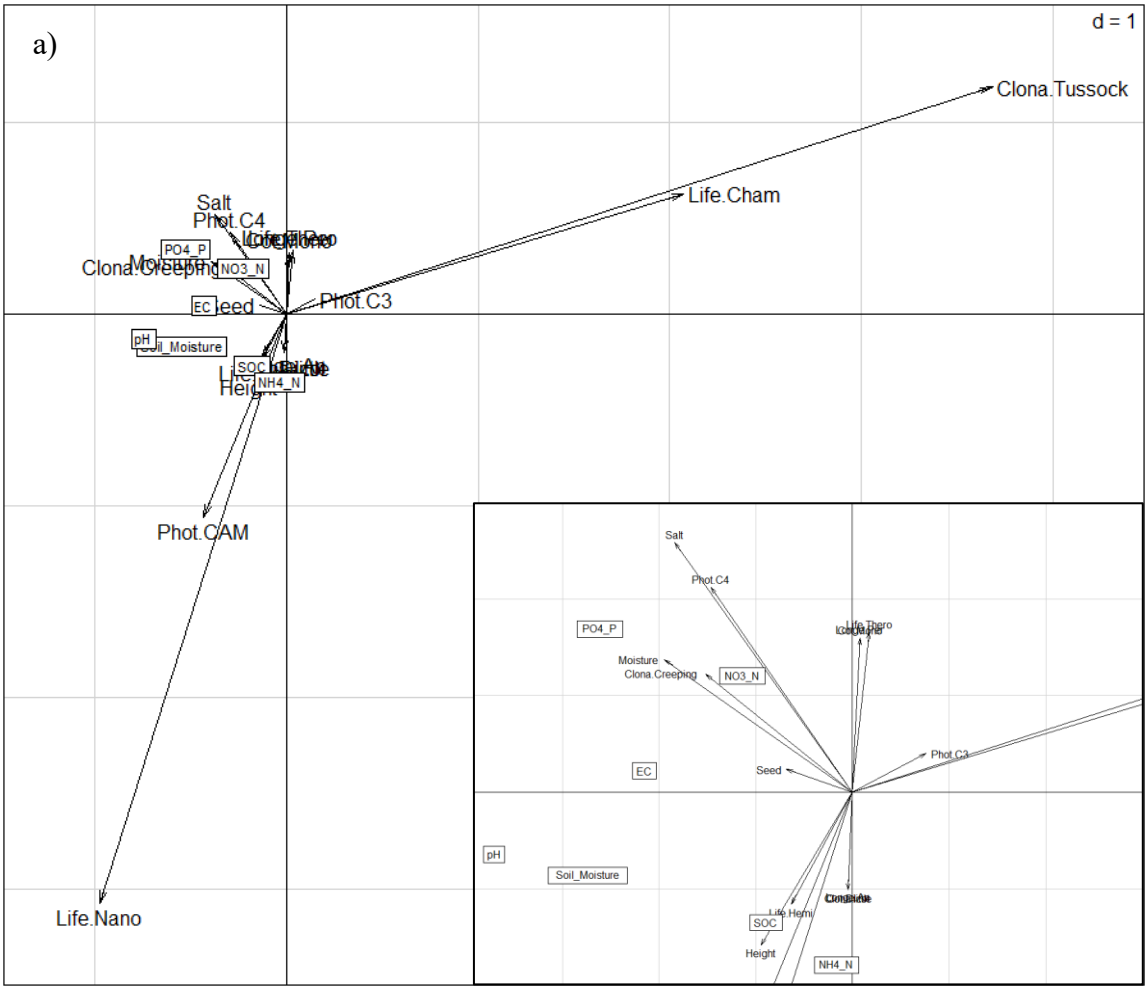
Plant trait	Axis1	Axis2
Clonality - creeping	-0.6763	0.7102
Clonality - little spread	0.1939	-0.3798
Clonality - tussock	3.9470	-0.6664
Height (max)	-0.3475	-0.1185
Longevity - Annual	0.1939	-0.3798
Longevity - Perennial	-0.3064	0.6001
Photosynthetic pathway - C3	0.4013	0.0428
Photosynthetic pathway - C4	-1.3483	1.1130
Photosynthetic pathway - CAM	-0.3190	-1.8406
Seed weight	-0.5250	0.4174
Cotyledon - Dicot	0.1939	-0.3798
Cotyledon - Monocot	-0.3064	0.6001
Life form - Chamaephyte	1.7729	0.1727
Life form - Hemicryptophyte	-0.2731	-0.0221
Life form - Nanophanerophyte	-0.5961	-2.8344
Life form - Therophyte	0.2532	0.0719
Salt tolerance	-0.6181	0.6309
Moisture tolerance	-0.8831	0.5483

The C3 photosynthetic pathway, tussock clonality, and chamaephyte life form traits were negatively correlated with all soil variables. Maximum height and the nanophanerophytes life form trait were negatively correlated with NO_3-N, and seed weight and moisture tolerance were negatively correlated with NH_4-N. The C4 pathway, salt tolerance and creeping (clonality) traits had a positive relationship with most variables apart from SOC and NH_4-N, whereas the CAM pathway and hemicryptophyte life form traits had a negative relationship with NO_3-N and PO_4-P, which was the opposite to the therophyte life form trait. The dicotyledon trait was positively correlated with moisture, pH, SOC and NH_4-N, which was the opposite to the monocotyledon trait (Table 6.5). However, the dicotyledon trait was analogous with the little spread (clonality) trait. The perennial longevity trait had a positive relationship with EC, NO_3-N, and PO_4-P, but was negatively correlated with all other traits (Table 6.5; Figure 6.9).

Table 6.5 Relationships between plant traits and C1 soil variables. Figures in red font are negative correlations.

Soil variable	Clonality creeping	Clonality Little spread	Clonality Tussock	Height (max)	Longevity Annual	Longevity Perennial
Moisture	0.1126	0.0432	-2.1479	0.1688	0.0432	-0.0683
pH	0.1695	0.0589	-3.1125	0.1416	0.0589	-0.0930
EC	0.1903	-0.0455	-1.2904	0.1104	-0.0455	0.0718
SOC	-0.0106	0.0568	-1.0011	0.1281	0.0568	-0.0898
NO_3-N	0.1077	-0.0247	-0.7507	-0.0529	-0.0247	0.0390
NH_4-N	-0.0572	0.0568	-0.4632	0.1290	0.0568	-0.0897
PO_4-P	0.2558	-0.0783	-1.3950	0.0072	-0.0783	0.1237
Soil variable	C3 pathway	C4 pathway	CAM pathway	Seed weight	Dicotyledon	Monocotyledon
Moisture	-0.1044	0.0717	0.4841	0.0576	0.0432	-0.0683
pH	-0.1118	0.1098	0.4708	0.0650	0.0589	-0.0930
EC	-0.0808	0.2300	0.1239	0.1040	-0.0455	0.0718
SOC	-0.0452	-0.0189	0.2811	0.0401	0.0568	-0.0898
NO_3-N	-0.0021	0.1178	-0.1573	0.0219	-0.0247	0.0390
NH_4-N	-0.0305	-0.1445	0.3792	-0.0118	0.0568	-0.0897
PO_4-P	-0.0527	0.3259	-0.1723	0.1036	-0.0783	0.1237
Soil variable	Life form Cham	Life form Hemi	Life form Nano	Life form Thero	Salt tolerance	Moisture tolerance
Moisture	-1.2868	0.1078	0.7392	-0.0645	0.1189	0.1747
pH	-1.5061	0.0978	0.7554	-0.0340	0.2168	0.2519
EC	-1.0633	0.1027	0.3830	-0.0678	0.1541	0.1692
SOC	-0.4339	0.0790	1.4289	-0.1036	-0.0834	0.0005
NO_3-N	-0.2460	-0.0501	-0.9615	0.1068	0.2212	0.1430
NH_4-N	-0.3860	0.0955	1.5687	-0.1328	-0.1686	-0.0971
PO_4-P	-0.8908	-0.0098	0.2689	0.0769	0.3665	0.3020

From the RLQ analysis, clustering using Euclidean distance showed the grouping of species in relation to soil environmental variables and species traits (Figure 6.10). The Calinsky-Harabasz criterion showed that there were two functional groups which were ascertained and plotted on the RLQ axis in relation to species traits and species positions.



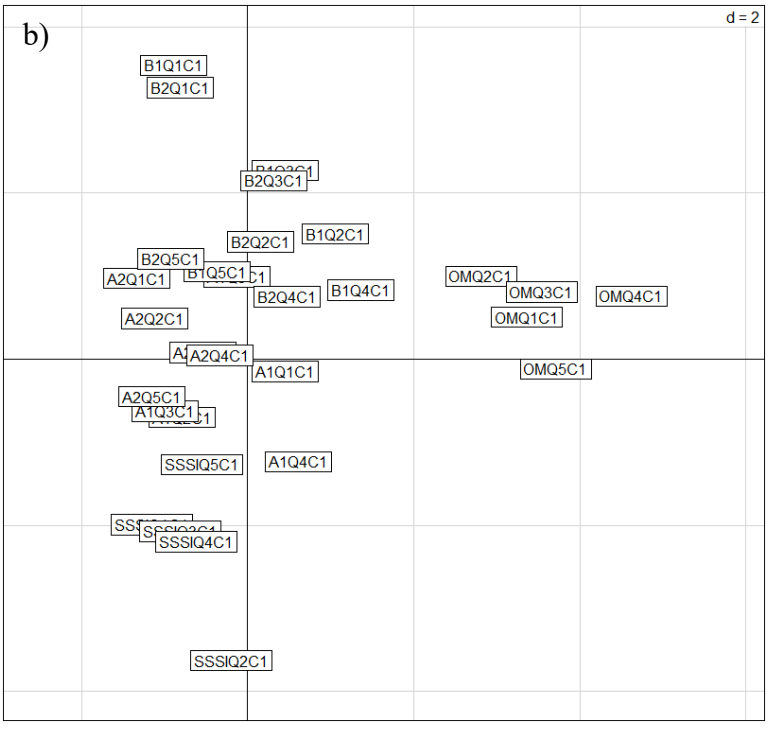


Figure 6.9 RLQ analysis ordinations of August 2017 data; a) traits and soil variables, inset is magnified image; b) Quadrats (C1).

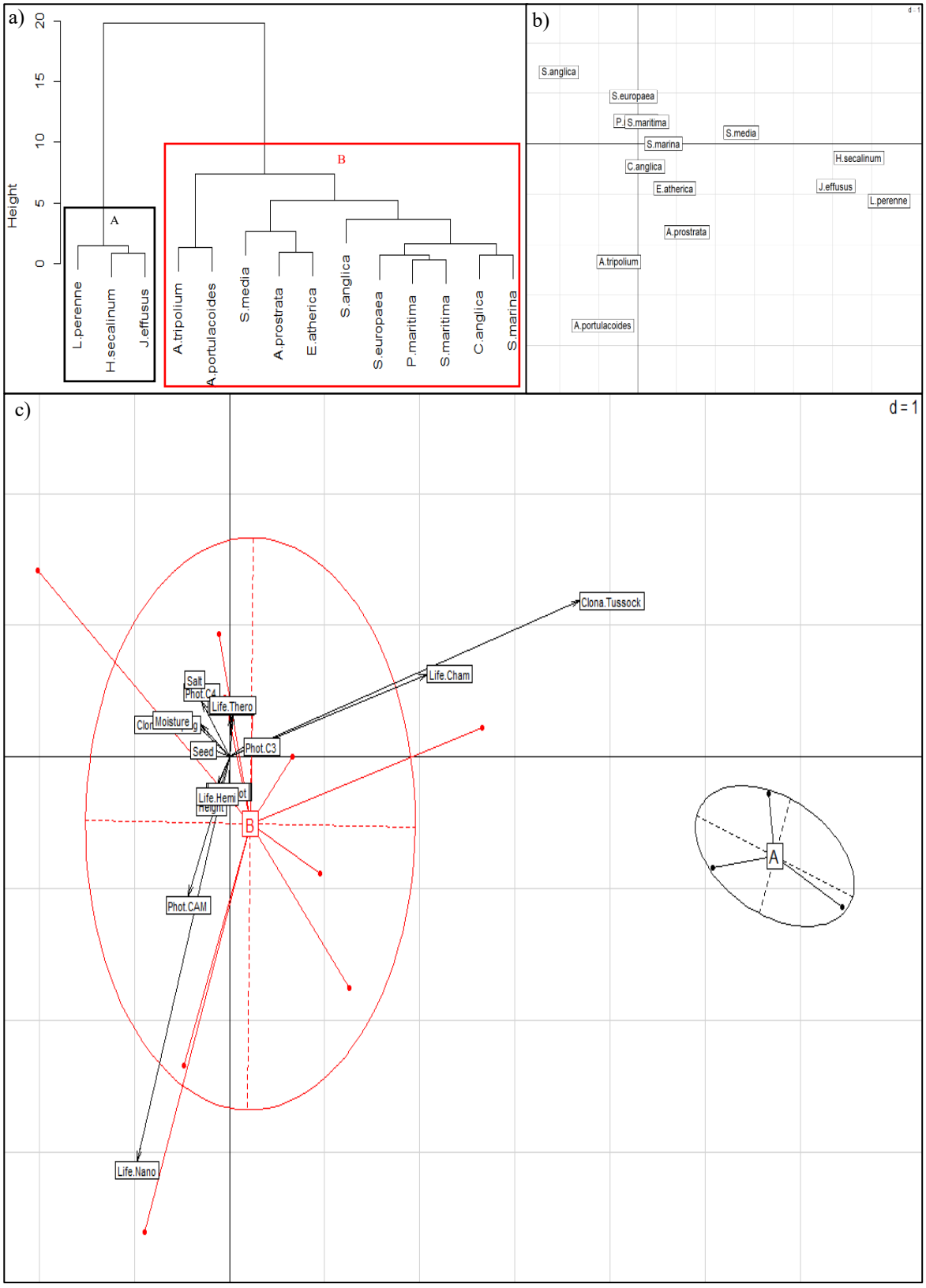


Figure 6.10 RLQ analysis ordination of species functional groups based on C1 soil variables; a) cluster dendrogam of species functional groups - functional group A is indicated by black outline, functional group B is indicated by red outline; b) ordination of species; c) functional groups in relation to species traits - points radiating from each functional group are the species in the top right ordination.

6.3.2 C2 cores

The first two axis in the RLQ analysis were responsible for 95.6% of the variation, but the first axis was the most meaningful (eigenvalue > 1). All soil variables had negative correlations with the first axis, and most variables had negative correlations with the second axis, apart from PO_4-P, which had a positive correlation > 0.7 (Table 6.6).

Table 6.6 C2 Soil variable correlations on RLQ axes. Figures in red font are negative correlations.

Soil variable	Axis1	Axis2
Soil moisture	-0.7910	-0.6935
pH	-0.7350	-0.2835
EC	-0.8467	-0.4914
SOC	-0.7445	-0.7809
NO3_N	-0.6439	-0.3947
NH4_N	-0.8275	-0.7363
PO4_P	-0.1020	0.7346

Most traits had a negative correlation with the first axis, apart from tussock clonality, the perennial longevity trait, C3 pathway, and the monocot, chamaephyte and therophyte traits. 50% of the traits were positively correlated with the second axis: creeping (clonality), height, perennial (longevity), C4 pathway, seed weight, monocot (cotyledon), hemicryptophyte (life form), salt tolerance and moisture tolerance (Table 6.7).

Table 6.7 Plant trait correlations on RLQ axes related to C2 soil variables and plant presence absence data. Figures in red font are negative correlations.

Plant trait	Axis1	Axis2
Clonality -creeping	-0.2874	1.1243
Clonality - little spread	-0.0578	-0.6193
Clonality -tussock	4.4471	-0.6974
Height (max)	-0.2036	0.4042
Longevity - Annual	-0.0578	-0.6193
Longevity - Perennial	0.0913	0.9785
Photosynthetic pathway -C3	0.3704	-0.2337
Photosynthetic pathway -C4	-0.8237	1.7751
Photosynthetic pathway - CAM	-0.8993	-1.2372
Seed weight	-0.2897	0.7478
Cotyledon - Dicot	-0.0578	-0.6193
Cotyledon - Monocot	0.0913	0.9785
Life form - Chamaephyte	1.9106	-0.5968
Life form - Hemicryptophyte	-0.1387	0.3653
Life form - Nanophanerophyte	-0.7969	-1.7986
Life form - Therophyte	0.0604	-0.4253
Salt tolerance	-0.5855	0.4298
Moisture tolerance	-0.7595	0.6892

The C3 photosynthetic pathway, tussock clonality, and chamaephyte life form traits were negatively correlated with all soil variables in C2 cores. Maximum height and the hemicryptophyte life form trait were negatively correlated with NO_3-N, but positively correlated with all other variables. Moisture tolerance was negatively correlated with SOC.

The CAM pathway, annual trait, little spread (clonality) and dicotyledon traits were negatively correlated with PO_4-P (Table 6.8; Figure 6.11).

Table 6.8 Relationships between plant traits and C2 soil variables. Figures in red font are negative correlations.

Soil variable	Clonality creeping	Clonality Little spread	Clonality Tussock	Height (max)	Longevity Annual	Longevity Perennial
Moisture	-0.0102	0.0629	-1.1241	0.1001	0.0629	-0.0993
pH	0.1499	0.0633	-2.9742	0.1415	0.0633	-0.1000
EC	0.0829	0.0349	-1.6431	0.1406	0.0349	-0.0552
SOC	-0.0511	0.0818	-1.0276	0.0783	0.0818	-0.1293
NO_3-N	-0.0064	0.0421	-0.7576	-0.0122	0.0421	-0.0665
NH_4-N	-0.0324	0.0849	-1.3038	0.0627	0.0849	-0.1341
PO_4-P	0.2467	-0.0953	-0.9549	0.0257	-0.0953	0.1506
Soil variable	C3 pathway	C4 pathway	CAM pathway	Seed weight	Dicotyledon	Monocotyledon
Moisture	-0.0624	-0.1037	0.5000	-0.0268	0.0629	-0.0993
pH	-0.1047	0.0975	0.4486	0.0644	0.0633	-0.1000
EC	-0.0985	0.1066	0.4007	0.0650	0.0349	-0.0552
SOC	-0.0289	-0.1548	0.3853	-0.0364	0.0818	-0.1293
NO_3-N	-0.0314	-0.0723	0.2805	-0.0547	0.0421	-0.0665
NH_4-N	-0.0360	-0.1715	0.4489	-0.0552	0.0849	-0.1341
PO_4-P	-0.0595	0.3468	-0.1639	0.1168	-0.0953	0.1506
Soil variable	Life form Cham	Life form Hemi	Life form Nano	Life form Thero	Salt tolerance	Moisture tolerance
Moisture	-0.7618	0.0620	-0.2579	-0.0223	-0.0037	0.0238
pH	-1.4653	0.0921	0.9648	-0.0334	0.1893	0.2279
EC	-0.9515	0.0665	1.0513	-0.0391	0.1095	0.1404
SOC	-0.6428	0.0276	1.0858	-0.0097	-0.0347	-0.0152
NO_3-N	-0.3678	-0.0391	-0.5102	0.0922	0.1315	0.0718
NH_4-N	-0.8568	0.0111	0.3195	0.0444	0.0444	0.0047
PO_4-P	-0.8590	0.0106	-1.1706	0.0740	0.2999	0.2559

Clustering using Euclidean distance showed the grouping of species in relation to soil environmental variables and species traits (Figure 6.12a). The Calinsky-Harabasz criterion showed that there were two functional groups (Figure 6.12c), which were ascertained and plotted on the RLQ axis in relation to species traits and species positions (Figure 6.12b).

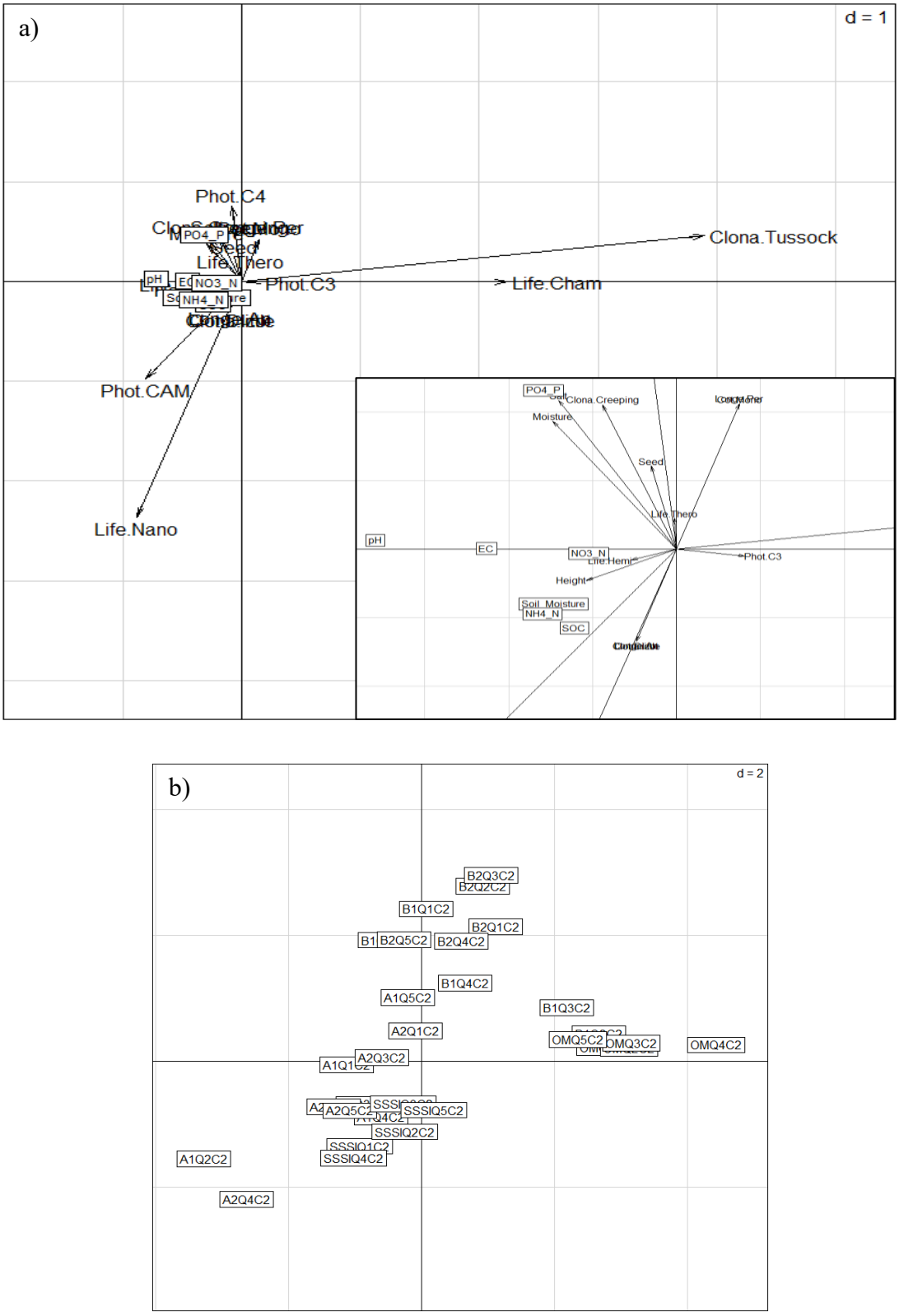


Figure 6.11 RLQ analysis ordinations of August 2017 data; a) traits and soil variables, inset is magnified image b); Quadrats (C2).

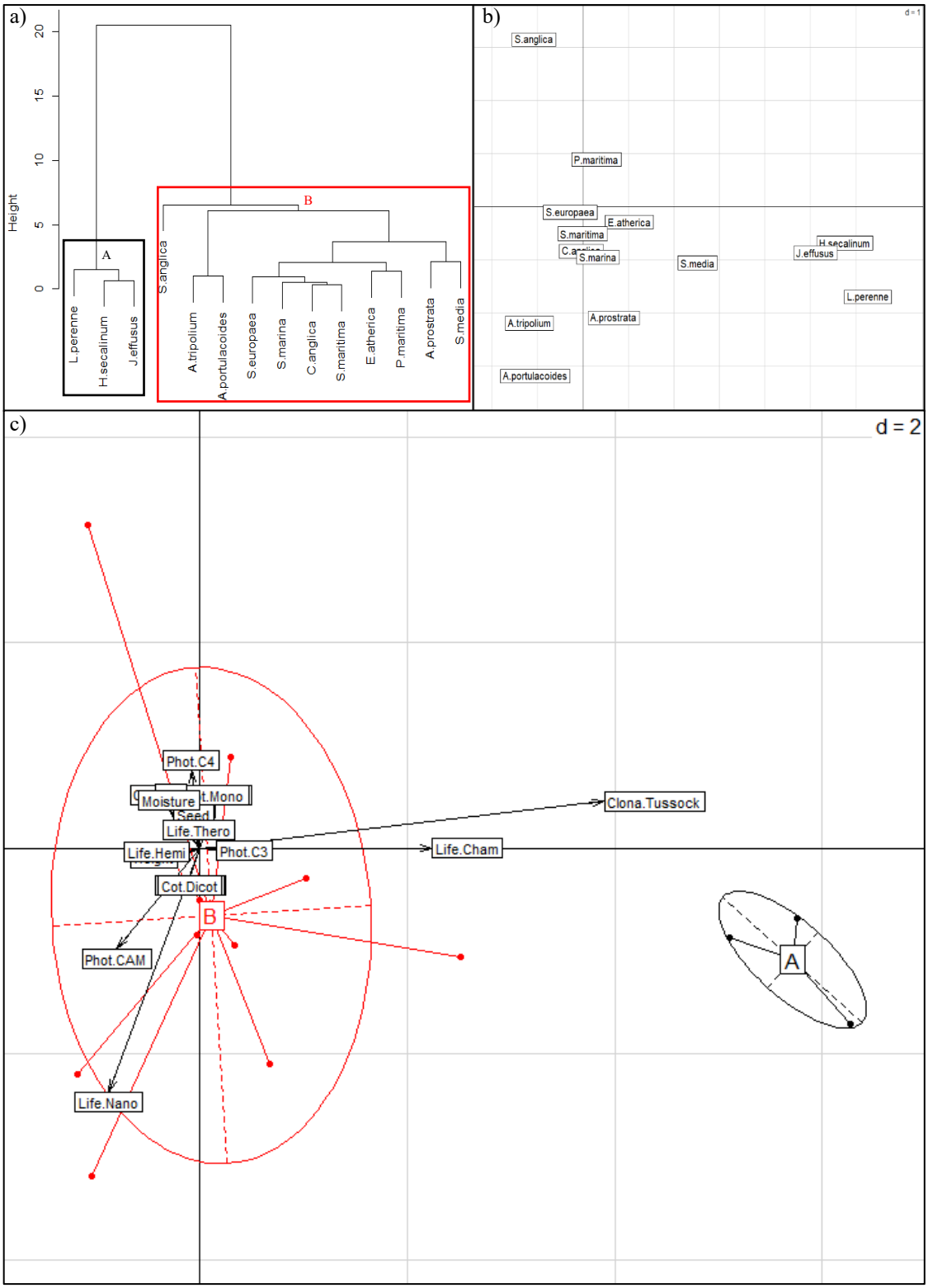


Figure 6.12 RLQ analysis ordination of plant species functional groups based on C2 soil variables; a) cluster dendrogam of species functional groups - functional group A is indicated by black outline, functional group B is indicated by red outline; b) ordination of species; c) functional groups in relation to species traits - points radiating from each functional group are the species in the top right ordination.

6.4 Influence of compaction and land height

The CCA model of August 2018 compaction, accretion and vegetation was significant (p = 0.001) and showed that 41% of the variation in vegetation was explained by compaction and land height. Variation on the first axis was non-random (p = 0.001). All other axes were not, although the second axis was close to significance (p = 0.065). Both compaction (p = 0.001) and AOD (p = 0.001) were highly significant after 999 permutations.

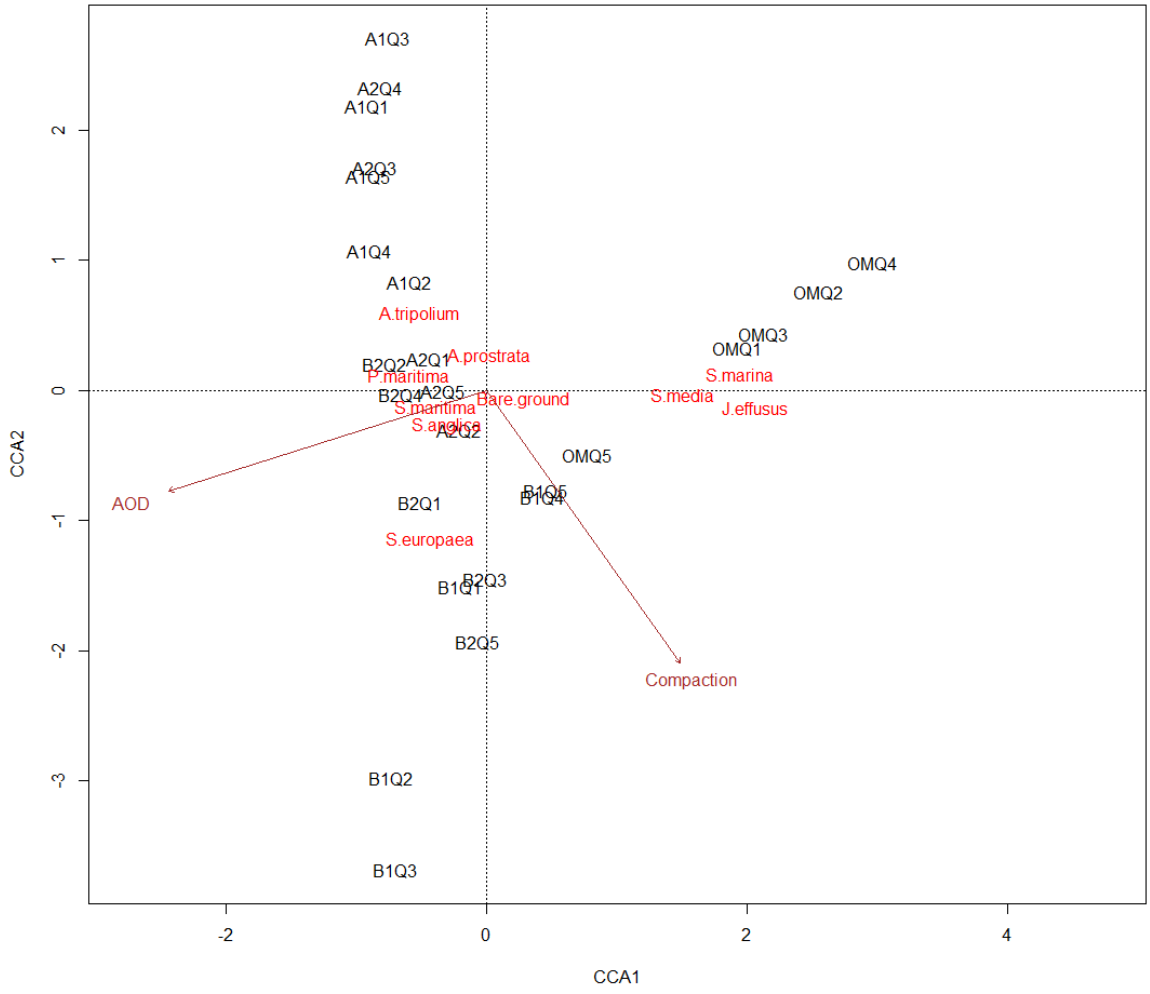


Figure 6.13 CCA ordination of August 2018 soil compaction and land height in relation to plant species composition and quadrats.

6.5 Ground-dwelling invertebrates

6.5.1 2014

The CCA model of August 2014 data (Figure 6.14) was significant (p = 0.001) and showed that 61% of the variation in ground-dwelling invertebrates was explained by soil variables. Variation on the first axis was non-random (p = 0.001), but no other axes significantly contributed to the ordination. Of the seven environmental variables in the

model, only soil moisture (p = 0.001), EC (p = 0.025) and NH_4-N were significant after 999 permutations. Site groupings in the CCA ordination show that sites were in similar positions to vegetation groupings in 2014, with terrestrial sites having negative scores on the first axis of ordination and the SSSI having positive scores, which reflects the differences between starting states.

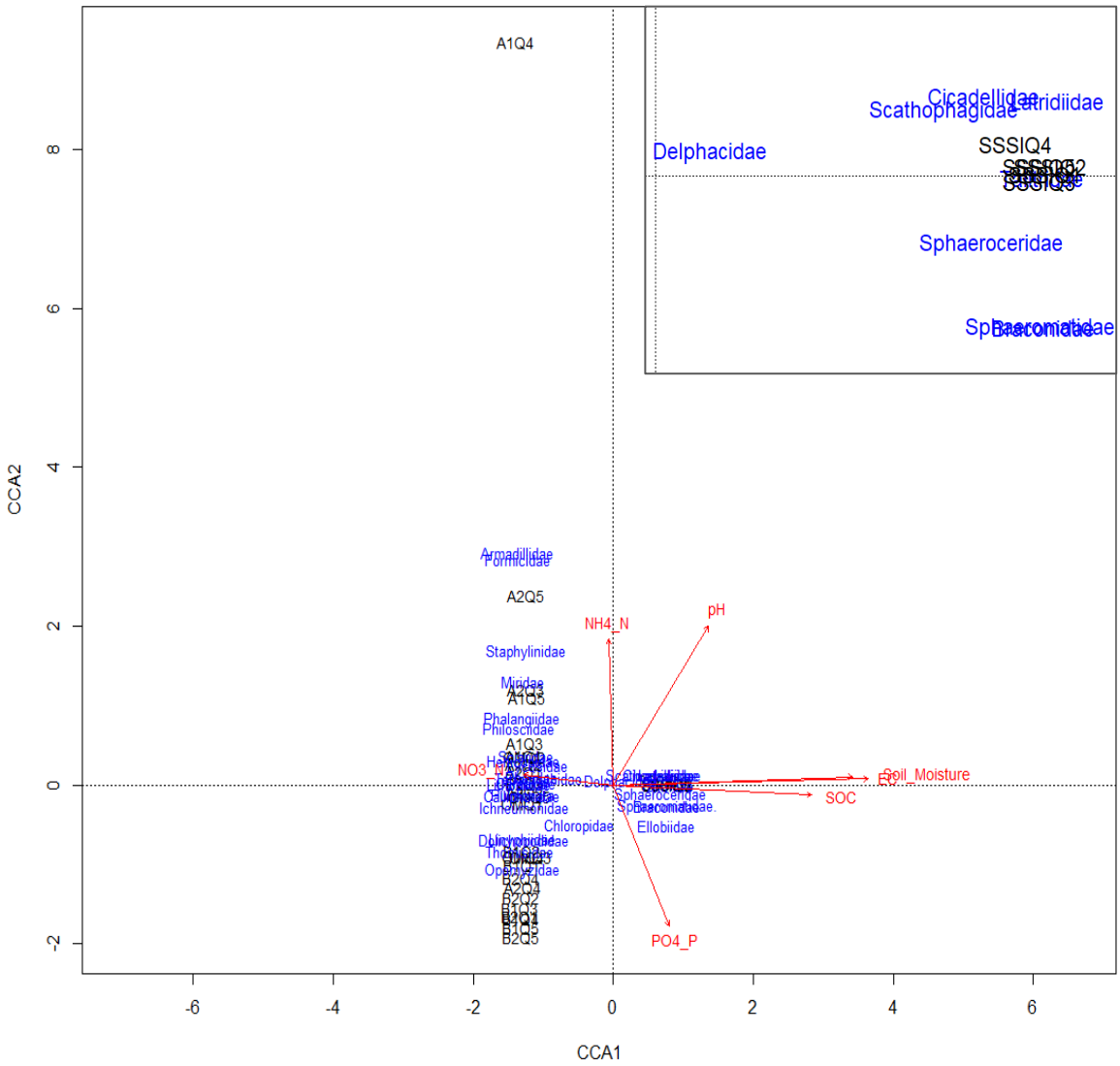


Figure 6.14 CCA ordination of ground-dwelling invertebrates and C1 soil variables in August 2014. Inset image is magnified.

6.5.2 2015

The CCA model of August 2015 data (Figure 6.15) was significant (p = 0.001) and showed that 54% of the variation in ground-dwelling invertebrate families was explained by soil variables. Variation on the first and second axes was non-random (p = 0.001; p = 0.028), and no other axes significantly contributed to the ordination. Soil moisture (p = 0.001), EC (p = 0.001) and NO_3-N (p = 0.001) were all significant after 999 permutations.

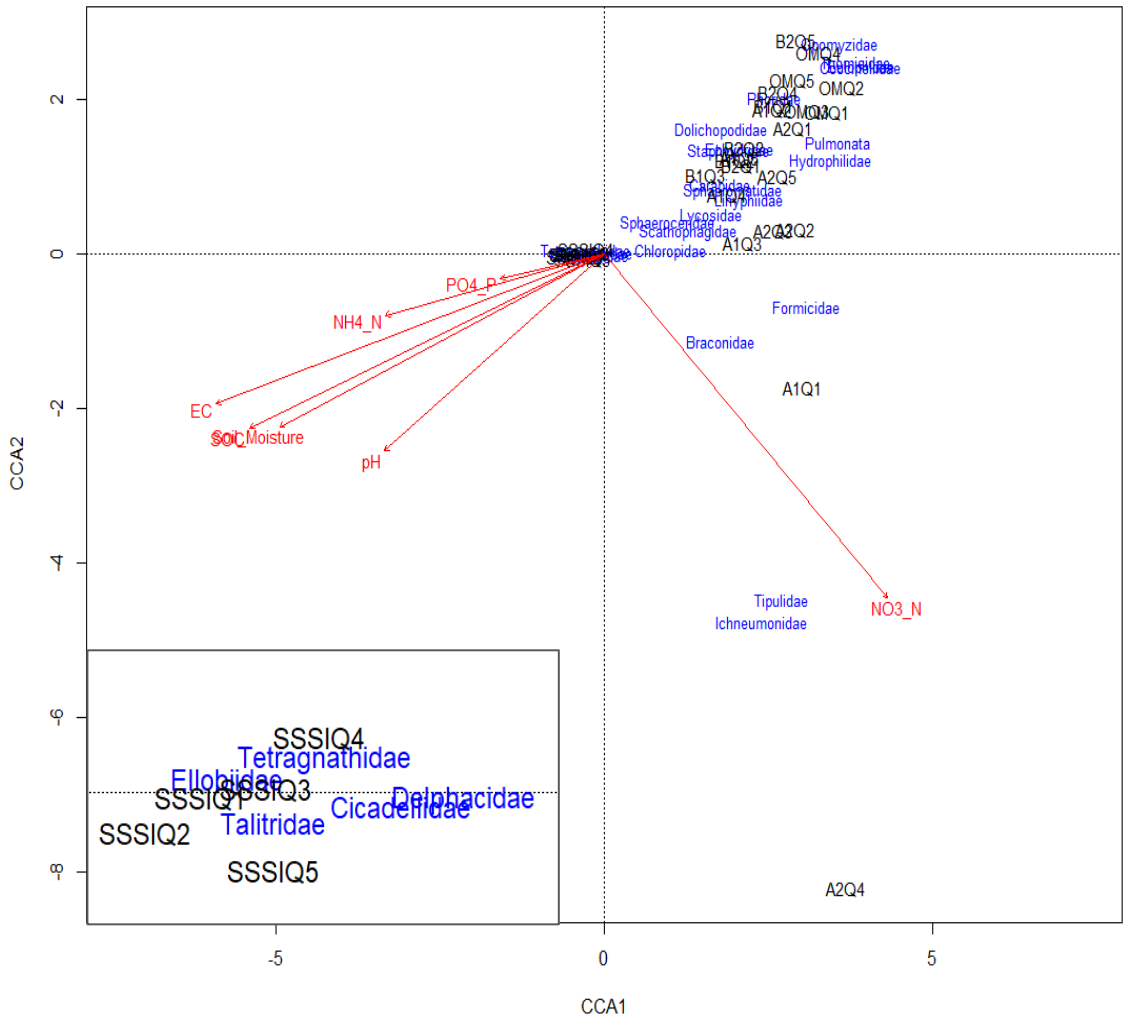


Figure 6.15 CCA ordination of ground-dwelling invertebrates and C1 soil variables in August 2015. Inset image is magnified.

6.5.3 2016

The CCA model of August 2016 data (Figure 6.16) was significant ($p = 0.006$) and showed that 36% of the variation in ground-dwelling invertebrate families was explained by soil variables. Variation on the first axis was non-random ($p = 0.001$), no other axes significantly contributed to the ordination. Soil moisture ($p = 0.001$), EC ($p = 0.024$) and SOC ($p = 0.025$) were significant after 999 permutations.

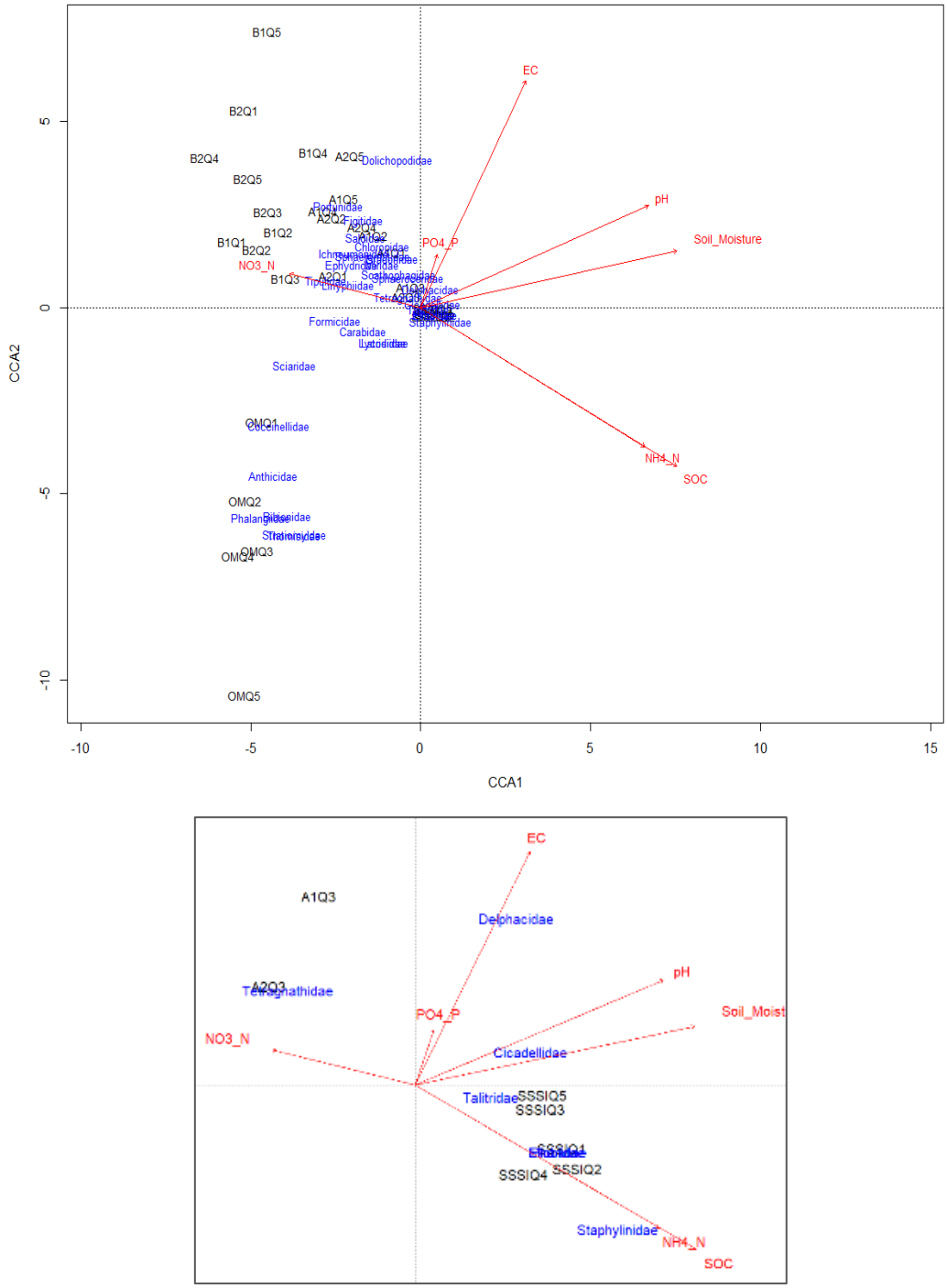


Figure 6.16 CCA ordination of ground-dwelling invertebrates and C1 soil variables in August 2016. Smaller image is the same ordination magnified.

6.5.4 2017

The CCA model of August 2017 data (Figure 6.17) was significant ($p = 0.001$) and showed that 44% of the variation in ground-dwelling invertebrate families was explained by soil variables. Variation on the first axis was non-random ($p = 0.001$), no other axes significantly contributed to the ordination. Soil moisture ($p = 0.001$), SOC ($p = 0.049$), NO_3-N ($p = 0.020$) and pH ($p = 0.036$) were significant after 999 permutations.

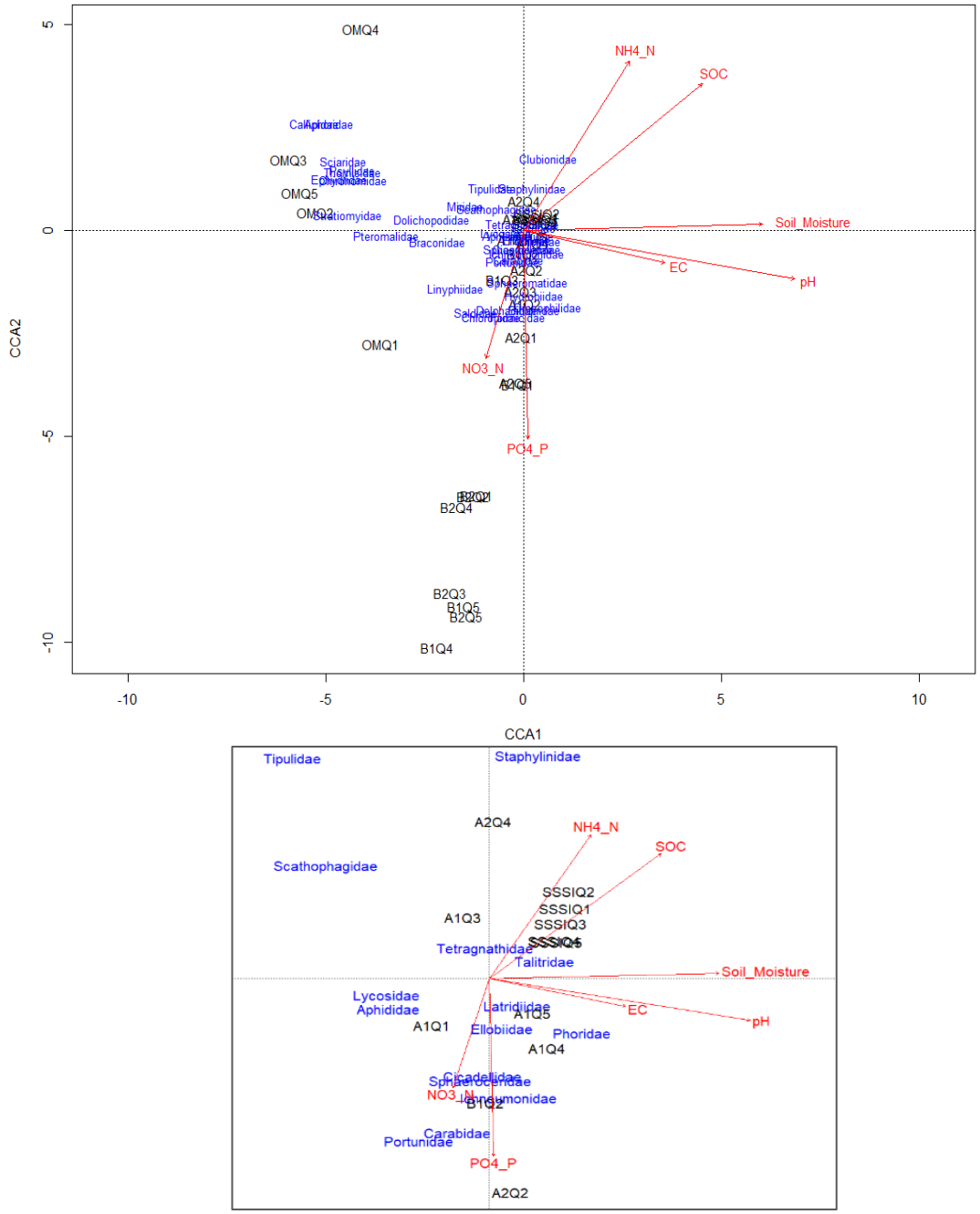


Figure 6.17 CCA ordination of ground-dwelling invertebrates and C1 soil variables in August 2017. Smaller image is the same ordination magnified.

6.6 Sweep net samples

6.6.1 2014

The CCA model of August 2014 data (Figure 6.18) was significant ($p = 0.001$) and showed that 34% of the variation in plant-dwelling invertebrate families was explained by soil variables. This was substantially lower than in the 2014 ground-dwelling invertebrate CCA model (61% of the variation), which was to be expected because the soil would have a direct influence on ground-dwelling invertebrates, and invertebrates caught in sweep nets

also include more mobile insects than soil dwelling species. Variation on the first axis was non-random (p = 0.001), but no other axes significantly contributed to the ordination. Soil moisture (p = 0.001) and EC (p = 0.027) were significant after 999 permutations, suggesting that these variables were having some influence on the groupings on the first axis.

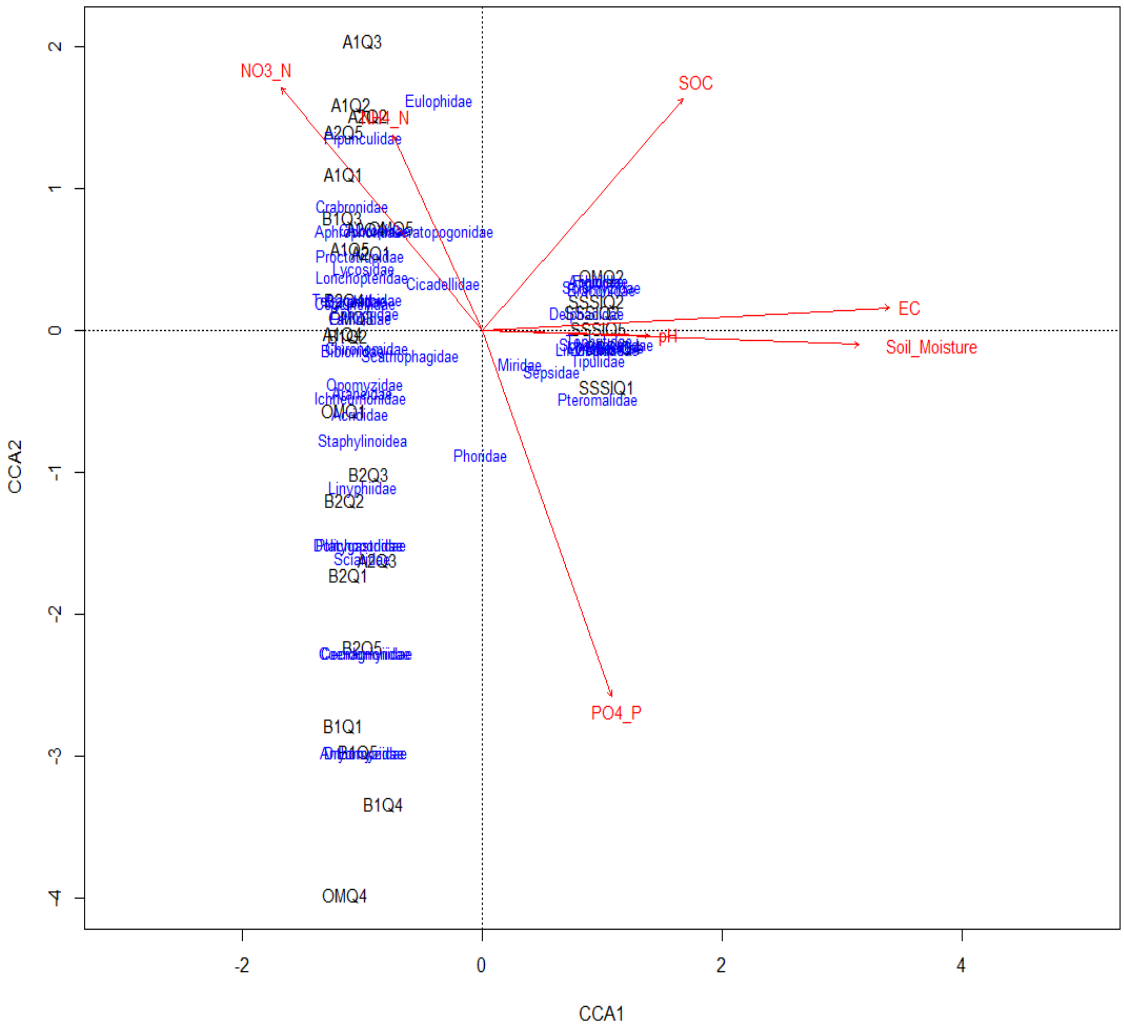


Figure 6.18 CCA ordination of plant-dwelling invertebrates and C1 soil variables in August 2014.

6.6.2 2015

The CCA model of August 2015 data (Figure 6.19) was significant (p = 0.001) and showed that 40% of the variation in plant-dwelling invertebrate families was explained by soil variables. Variation on the first axis was non-random (p = 0.001), but no other axes significantly contributed to the ordination. Soil moisture (p = 0.001) and EC (p = 0.018) were significant after 999 permutations. 50% of the Site A quadrats were in the closest proximity to the SSSI quadrats on the first axis.

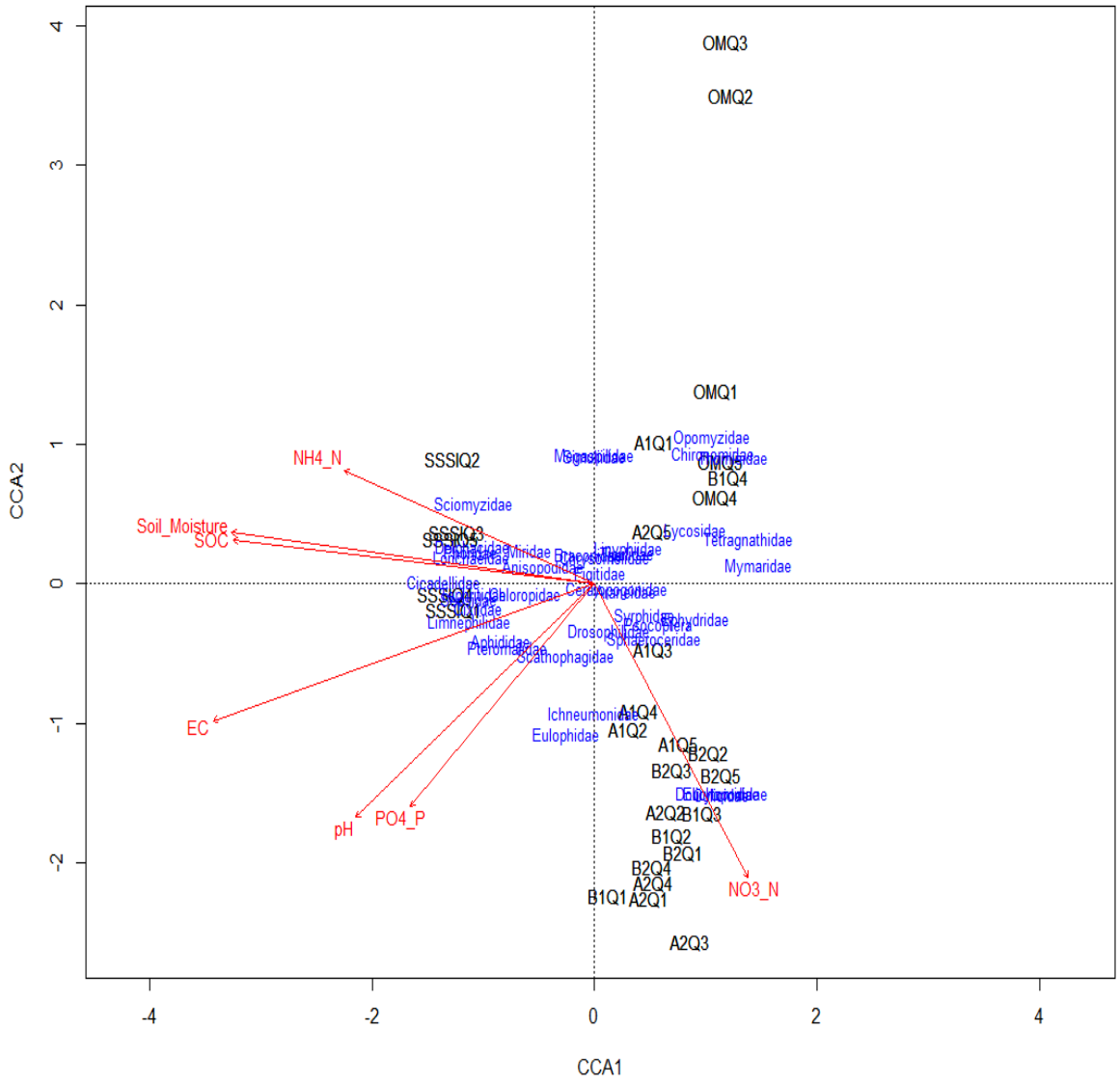


Figure 6.19 CCA ordination of plant-dwelling invertebrates and C1 soil variables in August 2015.

6.6.3 2016

The CCA model of August 2016 data (Figure 6.20) was significant ($p = 0.002$) and showed that 43% of the variation in plant-dwelling invertebrate families was explained by soil variables. Variation on the first axis was non-random ($p = 0.001$), but no other axes significantly contributed to the ordination. Soil moisture ($p = 0.001$) and SOC ($p = 0.002$) were significant after 999 permutations. By 2016 most of the Site A quadrats were closer than the other sites to the SSSI. This was based on invertebrate family assemblages and was primarily driven by soil moisture and SOC.

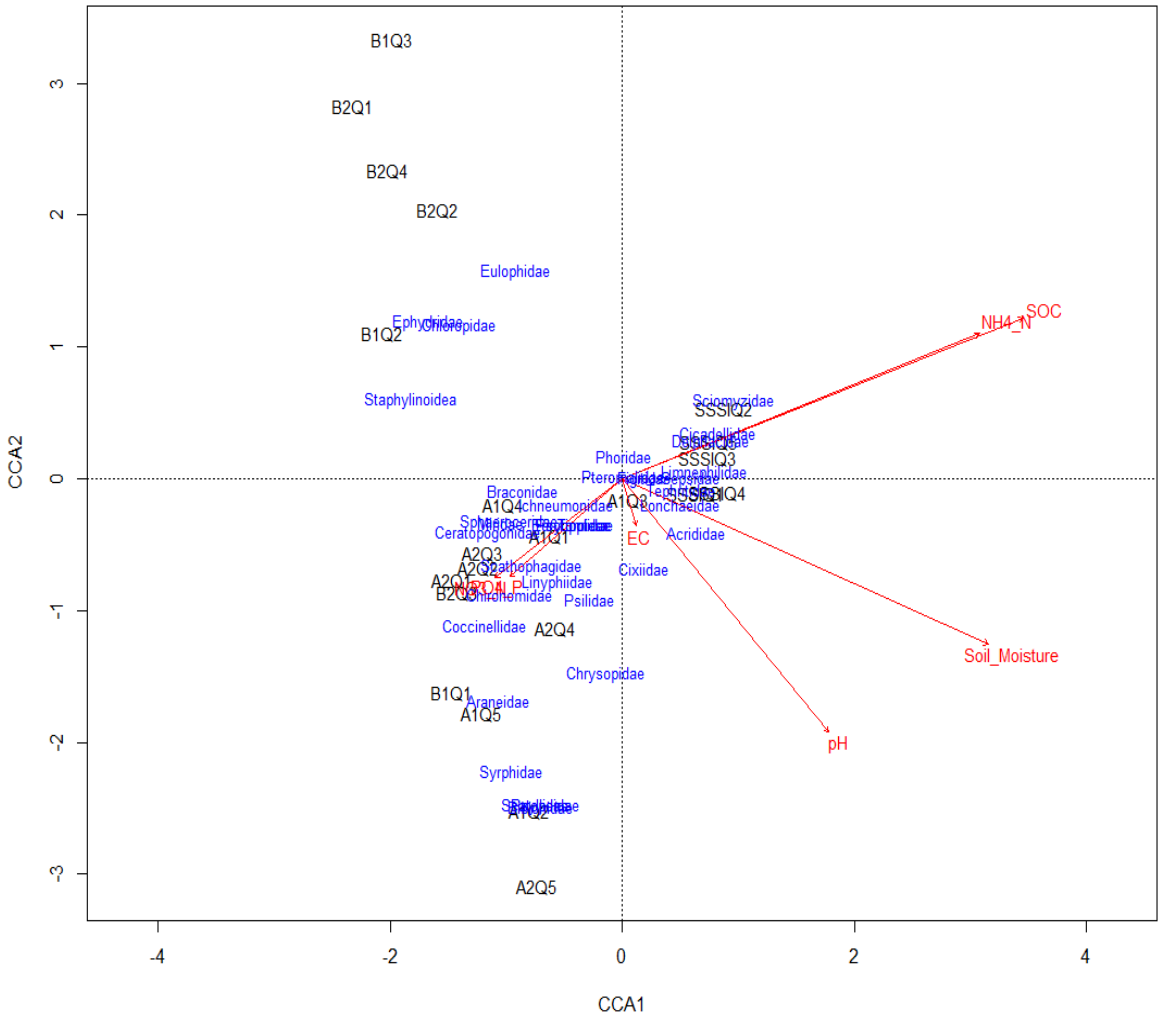


Figure 6.20 CCA ordination of plant-dwelling invertebrates and C1 soil variables in August 2016. OM was not included in the analysis because vegetation coverage was too low to use a sweep net.

6.6.4 2017

The CCA model of August 2017 data (Figure 6.21) was significant ($p = 0.011$) and showed that 39% of the variation in plant-dwelling invertebrate families was explained by soil variables. Variation on the first axis was non-random ($p = 0.025$), but all other axes were not significantly different to random. Soil moisture ($p = 0.001$) and pH ($p = 0.032$) were significant after 999 permutations. Many of the Site B quadrats had become closer to the Site A quadrats and the SSSI by 2017, which was based on these variables.

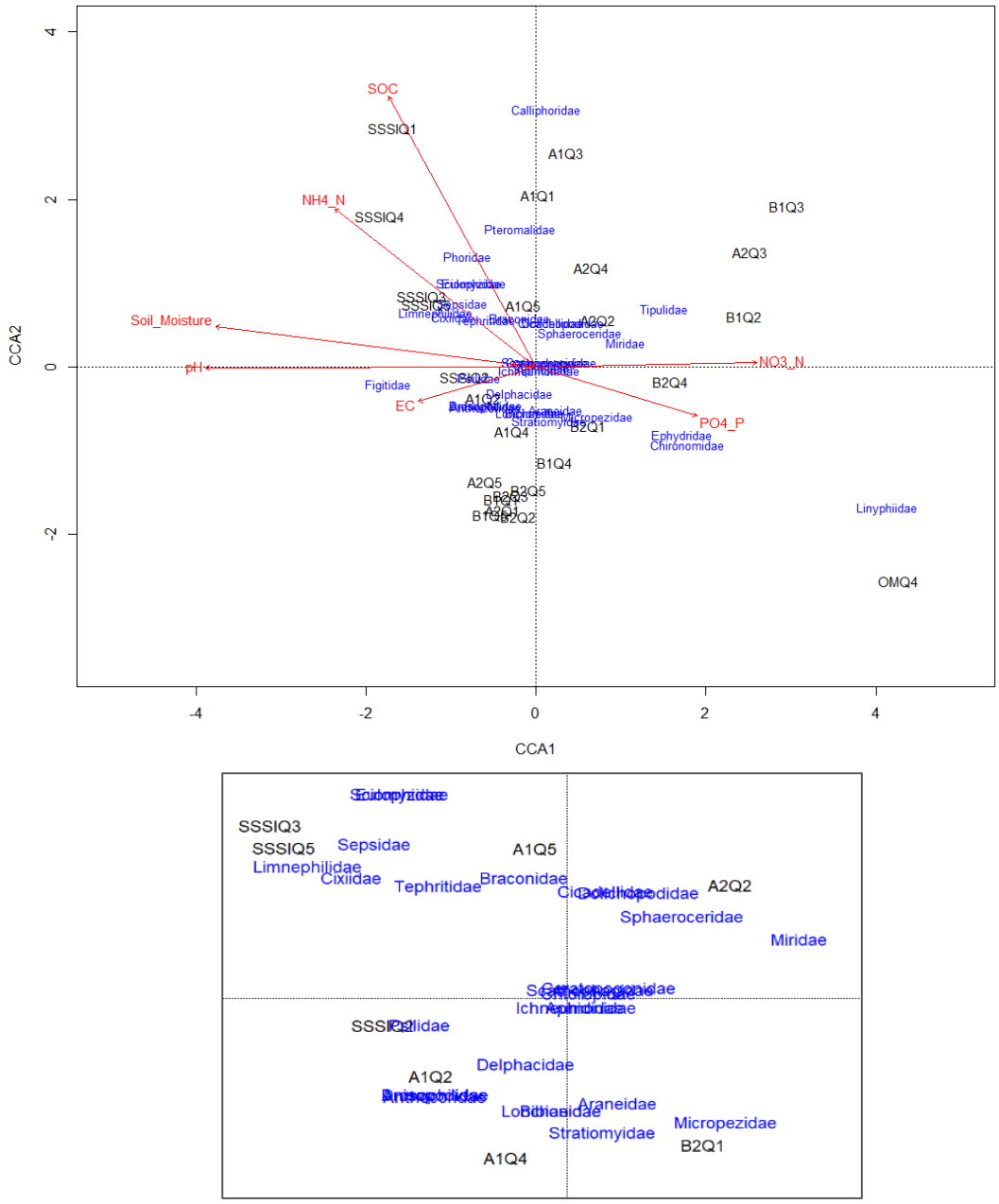


Figure 6.21 CCA ordination of plant-dwelling invertebrates and C1 soil variables in August 2017. Smaller image is same ordination magnified.

6.7 Soil-dwelling invertebrates

The proportion of inertia for the constrained variable in the soil-dwelling invertebrates CCA model for August 2017 data was 71%. However, the model (Figure 6.22) was not significant ($p = 0.506$). This is likely to be because numbers of invertebrates were very low, and sites had to be removed from the analysis due to this lack of data. C2 cores in relation to soil-dwelling invertebrates were also not analysed because numbers were lower than in the C1 cores.

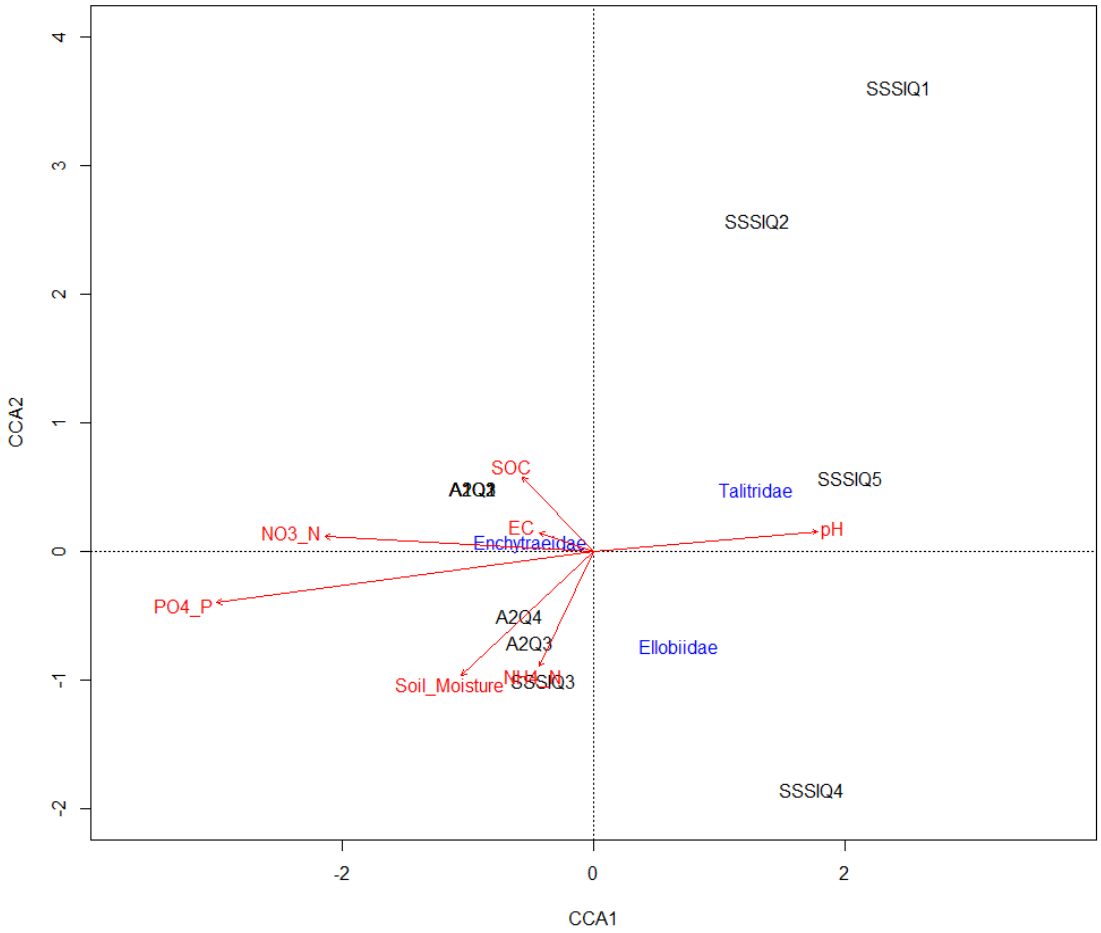


Figure 6.22 CCA ordination of ground-dwelling invertebrates and C1 soil variables in August 2017.

6.8 Non-metric multidimensional scaling

Non-metric multidimensional scaling using Euclidean distance (Figure 6.23) shows the grouping of quadrats when all August 2017 variables (vegetation, soil characteristics, ground-dwelling, plant-dwelling, and soil-dwelling invertebrates) were included in the analysis. Stress of the plot was 0.11 (11%) and the first axis had an R^2 value of 0.7, which is above the minimum acceptable level of 0.6. The second axis had an $R^2 < 0.6$, which suggests that the first axis is the most meaningful. 60% of the Site A plots were closer to three of the SSSI quadrats. SSSIQ1 was an outlier due to the high abundance of *O. gammarellus* in that quadrat, and because it was also the only quadrat to contain *A. portulacoides*. OMQ4 was an outlier on the opposite side of the NMDS plot because it contained different assemblages to other plots. A2Q1 was also an outlier, probably because a disproportionate number of Bibionidae flies were caught in the sweep net in that quadrat.

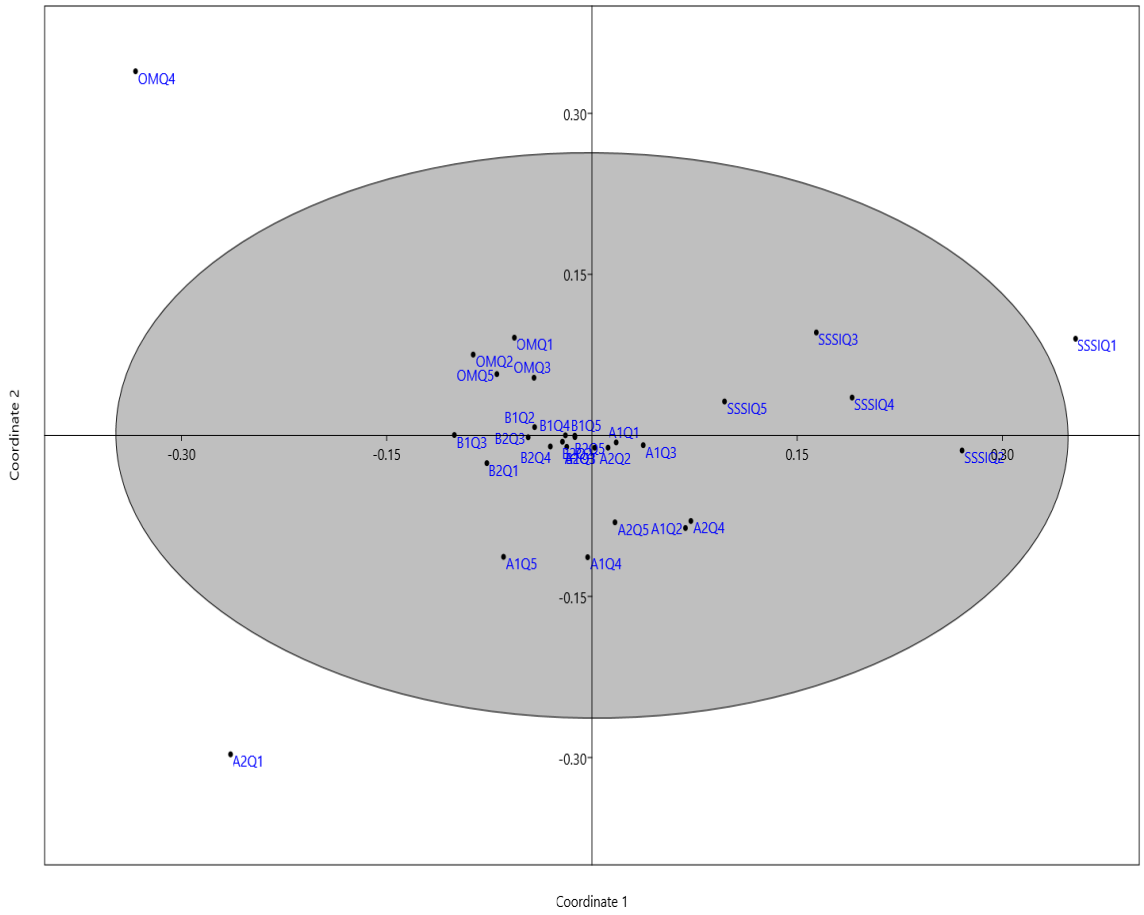


Figure 6.23 Non-metric multidimensional scaling ordination of all variables (vegetation, soil characteristics, ground-dwelling, plant-dwelling, and soil-dwelling invertebrates). Grey oval is 95% ellipses. Quadrats outside of the ellipses are outliers. Stress = 0.11, R^2 axis 1 = 0.7, and R^2 axis 2 = 0.5.

6.9 Discussion

In this chapter CCA analysis showed the groupings of quadrats related to vegetation and soil variables each year of the study. Patterns were similar to the groupings in the ordinations in Chapter 4 and showed that soil moisture and salinity (measured by electrical conductivity) were key influences on groupings of species and quadrats. By 2017 quadrats in Steart Marsh plots were closer to the SSSI on the first gradient, whereas OM was very different because of the lower moisture and salt content in the soil, and most other soil variables had lower measurements on OM. The higher phosphate and nitrate levels in C1 cores on the Site B plots in 2017 may have influenced the positioning of quadrats on the second axis, which were more distant to the SSSI than were the Site A quadrats.

Halophytes are defined as plant species that have adapted to grow in soils that are rich in salt (Allaby, 2004), so it was expected that EC levels in the soil would have an influence on the colonisation of saltmarsh species, with analysis showing that this was the case. Salt levels can range in saltmarsh sediments depending on how dry the soil is, and

vegetation does play a role in removing salt from the sediment (Moffett, Robinson and Gorelick, 2010). This may be why unvegetated quadrats on the Site B plots had particularly high levels of salt in the years following the breach. Soil moisture also had a major influence on site and plant species groupings, which has been found in other studies. For example, Schrama (2012) found that species richness of saltmarsh plants was higher when there was more moisture in the soil. Some species are more tolerant of waterlogging than others, such as *Salicornia* spp. which is tolerant of both salt and water (He, Altieri and Cui, 2015). *S. europaea* was more common on the Site B plots, and, although the soil was not necessarily more waterlogged on the Site B plots, it was under standing water for long periods. He, Altieri and Cui (2015) found that *Salicornia* spp. are commonly found near to pools of tidal water, which is similar to the distribution of *S. europaea* on the Site B plots.

RLQ analysis was used within this chapter to assess plants that had colonised Steart Marsh and OM in relation to the SSSI target state, and to soil variables and plant traits. Three of the plants (*J. effusus, H. secalinum and L. perenne*) found on OM in 2017 were remnants of what was on the site in 2014 and these formed functional group A (hereafter group A) in the analysis, whereas characteristic saltmarsh plants were in functional group B (hereafter group B). The plants in group A form tussocks, whereas all of the saltmarsh plants are either creeping or have little vegetative spread according to Fitter and Peat (1994). The three species in functional group A were all monocots, whereas some plants in group B were dicots. The only dicots found on OM in 2017 were *S. marina* and *S. media,* which are associated with coastal wetlands. Plants within each functional group were largely C3 plants, which was expected because most vegetation in temperate regions has a C3 photosynthetic pathway. However, there were two exceptions: *S. anglica* is a C4 plant and *A. tripolium* has a CAM pathway. These plants were both in functional group B, but these traits influenced their position in relation to the other plants in this group. *S. anglica* and *P. maritima* were the only monocots found on Steart Marsh and do compete with each other. They have different photosynthetic pathways, with *P. maritima* having a C3 pathway and *Spartina* is a C4 plant. This means that *P. maritima* has an advantage because it is better adapted to cooler, temperate climates. However, it is thought that *Spartina* could outcompete the species in the event of increasing temperatures and atmospheric CO_2 concentrations driven by climate change (Gray and Mogg, 2004).

Seed weight may have had an influence on seed dispersal on the MR site. *S. anglica* has the heaviest seed compared to all other plants, and this difference is reflected in the RLQ analysis. Seed weight may have an influence on saltmarsh plant dispersal (Wolters *et al.*,

2008), although flotation time of specific plant seeds may also have an effect, and this was not tested in this study. *S. anglica* was present on Steart Marsh in August 2015 (circa one year after the breach), but it was present in fewer quadrats than *P. maritima*, which is a monocot with lighter seeds, thus seed weight may have played a role in dispersal, because the lighter seeds are more likely to float and travel further. Both species are perennial and exhibit creeping clonality, and they can spread and outcompete annual plants. It is likely that their growth in some quadrats has reduced space for annuals such as *A. prostrata* to grow.

The RLQ analysis showed that the first axis was the most meaningful and it was related to tolerances of plants. The saltmarsh plants in group B are more tolerant of salt than the plants within group A, and most saltmarsh species are more tolerant of moisture, although *Juncus spp.* generally grow well in wet soils. *A. prostrata* is more tolerant to salt than the species in group A, but less tolerant than the other saltmarsh plants. However, as a therophyte it completes its lifecycle rapidly which compensates for salt stress. *E. atherica* is also less salt tolerant than other saltmarsh plants and it is usually found on upper marshes. It was found in only one quadrat on the SSSI and was not in any of the quadrats on Steart Marsh and OM. The SSSI plot was located further down the marsh than the swathes of *E. atherica* that were observed in the upper marsh on this site. This species is also less tolerant of waterlogging so its absence on the Steart Marsh plots and OM is not surprising. During construction of Steart Marsh, scrapes were created for birds, and *E. atherica* was observed on the islands in the middle of the scrapes post-breach. These islands were higher than surrounding areas.

Analysis showed that compaction and land height did influence the grouping of quadrats on Steart Marsh and OM. A decrease in soil aeration can reduce species richness on saltmarshes (Schrama, 2012), which suggests that this effect driven by compaction was having an influence on grouping in ordination, and the lack of soil-dwelling invertebrates in the soil may have also reduced aeration of the soil. The Site A plots were less compacted than the Site B plots and OM, and they were more vegetated than both sites. OM was marginally lower in height than the other plots, so submergence time could have been longer, which restricted growth. However, the Site B plots were higher AOD than the Site A plots. This could have meant that the Site B plots were less prone to submergence, but due to the compaction and height of the land surrounding this area Site B plots were under water for longer periods.

Saltmarsh plants are nitrogen limited (Mcfarlin *et al.*, 2008; Gedan, Silliman and Bertness, 2009; Vivanco, Irvine and Martiny, 2015), and nitrogen levels pre-breach were

higher on the MR than on the SSSI. This may have had a positive influence on the rapid colonisation of some halophytic plants, although nitrate levels were very variable across the study plots, especially in Site A, which suggests that the effect may not be apparent in every quadrat. Schrama (2012) suggests that the role of macro-detritivores in nitrogen mineralisation is likely to be beneficial to salt marsh plant growth in nitrogen limited systems. The macro-detritivore, *O. gammarellus* (family Talitridae), was the most abundant species on the SSSI and it exhibited a key role in the positioning of sites in ordination. *O. gammarellus* plays a substantial role in nutrient cycling and nitrogen mineralisation (Schrama, Berg and Olff, 2012; Schrama *et al.*, 2015), so it is reasonable to suggest this is why nitrate and SOC had an influence on the 2017 ordination. However, CCA analysis in 2017 showed that SOC and soil moisture had a greater influence on Talitridae and the SSSI quadrat ordination groupings than NO_3-N. Other invertebrates may have had an influence on decomposition. For example, Sphaeroceridae (lesser dung flies) were associated with NO_3-N in August 2017. Barnard (2011) states that species within this family are found on dung or rotting organic matter and they do not fly large distances, preferring to jump or run, which is why they were found in pitfall traps on the study sites. Sphaeroceridae spp. found on saltmarshes are macro-detritivores (Schrama, 2012), and thus have a role in breaking down organic matter. Soil moisture and EC also had a role to play in ground-dwelling invertebrate family composition, with increasing levels of these soil characteristics providing habitats for semi-aquatic halotolerant inveretebrate fauna, which compares to the effect that moisture and salinity had on plant presence.

As expected, the soil variables were responsible for a higher % of the variation of ground-dwelling invertebrates than of plant-dwelling invertebrates, because soil has a direct influence on the invertebrate fauna that lives on the ground. Nevertheless, there is an indirect association, because soil conditions also influence the growth of the plants species that plant dependent invertebrates use. Soil moisture and EC had some influence over the grouping of invertebrate families caught in sweep nets, possibly because the soil was appropriate for saltmarsh host plant species to establish, and it was predictable that lack of vegetation on some sites after the breach would have a negative impact on the presence and abundance of plant-dwelling invertebrate species. Some family groups appeared in different systems, and these could have been comprised of various species, such as Phoridae spp. (scuttle flies) which consist of 329 British species (Barnard, 2011), but in general there was a clear differentiation between plant-dwelling invertebrate families on the SSSI and the Steart Marsh plots and OM, which is likely to be due to the plant species found on the target state marsh. The plant-dwelling species described in Chapter 5, *C. plantaginis* (Tehritidae) and *P.*

marginata (Delphacidae), were having a similar influence on groupings in the August 2017 CCA analysis as they were in DCA ordination in the previous chapter.

Non-metric Multidimensional Scaling showed that most of the Site A quadrats were closer to SSSI quadrats when all variables were included in the analysis. There were outliers in this analysis, including one Site A quadrat, and one SSSI quadrat that had a high abundance of amphipods and a plant species that was found in no other quadrat. However, for the most part these data, along with the other analysis in this study, suggest that most Site A plots were similar to the target state three years after the breach.

To summarise, multivariate analysis has provided evidence that supports findings outlined in the previous three chapters. When all sampled variables were considered, more Site A quadrats were like the target state by 2017 compared to quadrats on all other plots. Soil moisture and EC (salinity) had key influences on the colonisation of plants and invertebrates on Steart Marsh and OM after the breach. Compaction and land height also influenced plant species composition and subsequent differences in plant coverage between sites.

CHAPTER 7: CONCLUSIONS

The aim of this study was to gather data that would help to assess the ecological changes that occurred following reconstruction and breaching of Steart Marsh in comparison to a managed brackish site with an agricultural starting state and a mature saltmarsh that was designated as a target state. The primary objective was to determine how plant, invertebrate and soil characteristics changed following a breach of existing sea defences, and to determine if there were differences in community changes between locations with different starting states. The expectation was that the new site would tend towards having community similarities with the mature saltmarsh adjacent to the reconstructed site.

The key finding in this study is that a clear pattern is apparent that shows that quadrats on the Steart Marsh MR in a site that was previously pasture were changing at a faster rate than a site that was previously arable in the direction of a SSSI saltmarsh target state. These sites were very different to a managed brackish site throughout the study. The pattern was apparent in terms of soil characteristics, vegetation and invertebrate colonisation and community structure. This may be due to compaction on the arable plots and associated anaerobic conditions, and the management and different conditions on the managed brackish site is likely to have had an influence on differences. This may have implications for the provision of ecosystem services by different areas across the scheme.

It was not clear at the start of this study which starting state would be the most appropriate for colonisation by saltmarsh plants and invertebrates, or if there would be differences in these processes in relation to the initial soil characteristics of the sites. However, data presented here indicates that changes post-breach differed between a site that was previously pasture (Site A), and a site that was formally arable (Site B), although this was not apparent for every quadrat within plots. Ecological characteristics on Site A and Site B were different year on year to an adjacent managed brackish site called Otterhampton Marsh.

The results of this research indicated that the pasture site changed toward the ecological state of the reference SSSI more rapidly than the other sites over the course of the study, indicating that this is a preferable starting state for rapid saltmarsh restoration or creation. However, the effects of compaction due to construction may be a confounding variable which makes it difficult to determine if the starting state is the governing factor. Moreover, although these sites are as comparable as possible geographically, the pasture site is physically closer to the SSSI, which may be one reason why the site transitioned quicker

as colonisation might have been more rapid. To further assess if pasture is more likely to rapidly change to saltmarsh on MR schemes, it is recommended that multiple sites with this starting state are monitored. The influence of distance can be assessed if studied pasture sites on MR schemes are selected at a range of distances from established seed banks.

It was clear that compaction did have an influence on the development of the arable site. These compacted Site B plots were lower in elevation than the surrounding area, and they had little growth compared to the Site A plots. Although the level of compaction was not tested empirically until 2018, the site had been regularly traversed during development pre-breach with heavy machinery, and it is likely that this could have had a major impact on plant growth and community composition in certain areas. The sediment arriving from the estuary accreted on these sites, which provided substrate for growth, but it is likely that the compaction further down the soil profile also had an influence on the hydrology (Spencer, 2017), causing water to remain on these locations. It is recommended that compaction should be a factor that is carefully considered during saltmarsh restoration, because it does appear to have an influence on succession in the early stages of saltmarsh development. It is evident from observations during the engineering phase that part of this compaction could have been caused by machinery during the construction phase, although arable soils can become compacted as a result of farming techniques (Hamza and Anderson, 2005). This may have been the case on Site B because the land had been in an arable crop rotation for many years prior to development of the MR scheme.

Although starting state and compaction have been identified as important determinants of how succession has proceeded, this study has not fully identified their relative importance. Further research on this is needed, ideally with replicated plots of each starting state with differing levels of compaction. This is important because heavy compaction resulting from site engineering during MR may influence succession and final functionality of re-created saltmarshes.

In this study, functional traits of plants were gathered from a British plant database. This was useful information because it showed functioning of plant communities across the study plots, and it was apparent that the saltmarsh plants on Steart formed a functional group that was different from plants still growing on OM by the end of the study. However, it would also be beneficial to measure traits in-situ on Steart Marsh to assess if there are differences between traits recorded on databases and traits of plants growing on MR sites. A comparison of specific traits found on the MR site could then be compared with plant traits of natural marsh species to determine if there are differences between plants traits in the two

different treatments. This will be valuable information because it will provide evidence of how plants are distributing and growing on the MR site in relation to known traits, and whether local conditions on the MR are suitable to facilitate similar growth and species composition and structure on the restored site.

Permanent quadrats with comparable heights were selected in this study to allow for a direct comparison between quadrats, which would theoretically have the same tidal regime. Permanent quadrats also provided evidence of succession in the same location each year. Additionally, the comparability of heights provided a good comparison with the SSSI target state, allowing for robust conclusions to be drawn from ordination and multivariate techniques. However, it was difficult to select sites that were exactly the same height AOD over such a large geographical area. It was also important that these sites would be accessible after the breach, and that the sites used did not conflict with the site requirements of WWT, which meant that selection of sites with suitable heights was limited. The permanent quadrats on all sites were set up within 0.5 metres in height of one another, but elevation does have a major influence on plant zonation on saltmarshes, and changes in elevation does have an effect (Mossman, Davy and Grant, 2012). This study was assessing saltmarsh flora and fauna in a lower marsh zone but does not consider the upper saltmarsh. Future work should consider locating study plots at different elevations on each site to gain an understanding of ecological patterns in different saltmarsh plant zones.

Early colonisers on the Steart Marsh plots were *A. prostrata* and *S. maritima*. These are both annual species and differences in the morphology of individual *A. prostrata* plants was observed, particularly in the first year after the breach, with some plants growing small leaves and others growing taller with large hastate leaves, perhaps due to differences in nutrients in the soil and variations in competition between plant species. *A. prostrata* and *S. maritima* were found on the SSSI plot and it was highly probable that the SSSI was facilitating colonisation on Steart. Both species died back over the winter of the first year and abundances dropped as perennial grasses began to colonise. *A. tripolium* was found in abundance on the SSSI and colonised the Site A plots. It was not in abundance on the Site B plots, although individual plants were noticed between the sample plots. S*partina anglica* colonised Steart Marsh and it will stabilise marsh sediment that has accreted substantially on the site. However, it may also reduce feeding grounds for wildfowl if it continues to grow and spread across the site. It is likely to continue to spread across Steart and will be difficult for managers to eradicate if it becomes detrimental, and its spread could be exacerbated by climate change. If temperatures continue to rise it could outcompete C3 species such as *P.*

maritima. Future research on the marsh should consider the effect of *S. anglica* on bird numbers.

A. portulacoides and *E. atherica* had low coverage on the SSSI plot, but these species were not recorded on Steart Marsh. The absence of *E. atherica* in the plots was not unexpected because it prefers higher elevations and it was seen on higher ground on Steart Marsh. The absence of *A. portulacoides* is more unexpected, because it inhabits the lower saltmarsh. It is not clear why this species did not colonise the sample plots during this study. However, it was encouraging that most of the plants found on the SSSI did colonise Steart Marsh, although coverage was not evenly spread across all quadrats.

There were still areas of bare ground on Steart Marsh by the end of the study. This is not like the SSSI plot, where all quadrats were fully vegetated. Evidence within the current decade suggests that bare ground is likely to persist, because this has occurred on other MR schemes (e.g. Davy *et al.* 2011; Brooks *et al.* 2015), but it is likely to decrease further with time (Mossman, Davy and Grant, 2012). It will be of interest to revisit the sites in the next few years to see if bare ground has reduced, especially on the Site B plots where it was most prevalent on Steart Marsh. Planting or seeding does not generally occur on British MR schemes (Sullivan *et al.*, 2018), but this could be considered if bare sediment persists in areas where it is not desired or if characteristic species are missing from communities, such as sea lavender (*Limonium vulgare*) and sea purslane (*Atriplex portulacoides*). Topography may also be an issue, because Site B plots were submerged by water for prolonged periods, and a recent suggestion has been made to increase topographical heterogeneity on MR sites so that they are more comparable to natural marshes (Lawrence *et al.* 2018).

Not all plots were accessible during every sampling period due to breeding birds on the sites. It was a condition that the research did not have a negative impact on nesting birds, and this was adhered to throughout the study. Fortunately, this inaccessibility only influenced vegetation sampling during the spring because most bird species were only breeding during this time. Vegetation data collected in spring were used to monitor intermediate change between August sampling points and did not affect any of the multivariate analysis in this study. Therefore, the issue with inaccessibility did not have a major influence on the conclusions that have been drawn.

During saltmarsh creation on Steart, physical features such as hedgerows were removed, but this is not usually the case on accidently breached sites. In future work, the ecology of accidently breached sites could be compared with MR schemes such as Steart

Marsh to assess if it is necessary to remove features of past-land use. An accidental breach occurred in the mid-90s in Porlock, Somerset and a policy of limited intervention was adopted. The saltmarsh is located circa 30 miles further west of Steart Marsh on the Severn Estuary, and could be used as a reference site due to this different management approach. This site still has remnants of woody plants, such as dead trees standing amongst the saltmarsh flora. Mossman, Davy and Grant (2012) found that plant species composition on accidently realigned saltmarshes is more similar to natural marshes than to MR sites, but this could be related to the age of the site because accidently realigned sites tend to be older. However, due to this evidence, it could be argued that heavy engineering may not always be needed to construct suitable habitats, although this may be required on MR schemes to construct the heterogeneity recommended by Lawrence *et al.* (2018). Although accidently breached sites appear to be more similar to natural marshes, they tend to have a greater abundance of *A. portulacoides* (Mossman, Davy and Grant, 2012). Anecdotal and pictorial evidence suggest that this species is abundant at Porlock, and salt pans have also formed on this site, so it would be of benefit to assess what the differences and similarities are across two differently managed sites on the Severn Estuary. Direct comparisons are difficult due to the differences in age between older accidental realigned sites and managed realigned sites, but data such as these can shed light on ecological differences that could aid creation of MR sites, and subsequent management decisions could be informed by these data.

Livestock grazing did occur on Steart Marsh during the study, but stocking density was low. Poaching of sediment and grazing of plants was noticed on the study sites. The most obvious evidence of poaching was on Otterhampton Marsh. Forty to fifty sheep grazed this site in 2015, and there were 14 yearling cattle on OM in 2016. In 2017, nine Dexter cattle grazed the site. Although stocking densities were not especially high, there were clear signs that cattle had traversed the quadrats, such as poaching and faeces in and around each quadrat. Poaching of the soil on OM may have had an impact on the erosion that was recorded from 2014 to 2018. Areas could be cordoned off on the marsh to compare grazed and non-grazed areas, because grazing can have both positive and negative effects. However, the stocking density of cattle on the marsh was below the maximum density recommended by Kleyer, Feddersen and Bockholt (2003), so negative effects are likely to have been minimal across the whole site. It would be interesting to assess the site usage of the cattle to see if they have preferences for certain areas on the site, because this could have an influence on colonisation. Cattle destroyed plot marker posts on all plots on Steart Marsh and OM, and it was initially planned that these would be used to measure accretion. This was mitigated by taking elevation measurements using an automatic level. It was also apparent that this

method was more accurate than measuring posts and allowed for a direct comparison with elevation data pre-breach.

The pitfall trap method adopted in this study worked effectively because windows between tides were carefully selected, and the traps were placed in the same position each year to gain an accurate comparison. Catches in pitfall traps were substantial, especially on the SSSI. However, there was an issue with pitfall traps flooding out on two separate occasions due to heavy rain in August 2015. The majority of washed out traps were on the Site B plots, which was under standing water, so it is unlikely that the loss of traps that year would have had a major effect on the overall findings. Cattle destroyed traps on Otterhampton in August 2017, which meant that these had to be reset in September of that year once cattle were removed from the site. This is likely to have had a minor influence on the catch on OM that year.

The most prevalent species found in the pitfall traps post-breach in this study was *O. gammarellus.* These were in high abundance on the SSSI and did begin to colonise Steart Marsh. The highest numbers on Steart Marsh were on the Site A plots, and they were absent from OM. This species was found on the surface of the sediment, but it was also found in some soil cores because it buries into the soil. The presence of this species is beneficial because it has a key role is soil aeration and evidence suggests that it is also involved in nitrogen mineralisation (e.g. Schrama, 2012; Schrama *et al.* 2015).

Apart from the amphipods caught in pitfall traps, soil-dwelling invertebrates were not abundant in soil cores during this study and were not representative of the type of fauna that was expected and found in other schemes, such as Hydrobia snails and Polychaete ragworms (Atkinson *et al.*, 2004; Mazik *et al.*, 2007). This lack of faunal richness may be due to the influence of construction pre-breach (compaction) and lack of vegetation post-breach. However, numbers were also low on the SSSI. The method of collecting soil cores and sifting through samples was tested in a neighbouring field and the abundance of invertebrates was higher, so there is merit to this method. However, it may be beneficial to test a range of other methods to assess soil invertebrates on a saltmarsh and MR site, such as sieving through soil cores with water. Low numbers of soil-dwelling invertebrates may have an influence in soil dynamics and the brown-food web, but the lack of these invertebrates may also be problematic at higher trophic levels.

The most abundant invertebrate families caught in sweep nets across all sites by the end of the study were Delphacidae and Tephritidae. The species within these families were

identified as *Prokelisia marginata* and *Campiglossa plantaginis* respectively. The former is an invasive species that is associated with *S. anglica*, and it is likely to spread further in parallel with this plant. Predation by other species found on the study sites, such as Lycosidae spiders, is unlikely to effectively limit this spread, and control methods for *Spartina* are difficult. *C. plantaginis* abundance may have been affected by cattle grazing, because many of the sea aster plants were eaten, but this did not seem to be the case. The reduction of sea aster on the sites can be seen as a negative because the plants support herbivorous invertebrates (Nolte, Esselink and Bakker, 2013), which are important food sources for birds.

In this study invertebrates were identified to family level, and three of the most prevalent species across all sites were identified to species. This gave a good indication of the families of invertebrates that colonised each year and indicated which species were dominant on the ground and living on plants across all the study sites. This is important information because the invertebrate fauna has a key role to play in productivity and food webs. Future long-term studies on Steart Marsh could specifically focus on the invertebrate fauna. The research could be dedicated to identifying invertebrates to species level where possible, and traits could be researched from the literature. From this information functional groups could be calculated using RLQ analysis, which would provide further evidence of how the MR is functioning in comparison to reference sites.

The seven soil variables assessed in this study provided a good representation of variables that were likely to influence the development of saltmarsh flora and fauna on Steart Marsh and OM. Soil moisture and salt had an impact on succession, which was expected because these are two key components of saltmarsh ecosystems. Phosphate levels were highest on the Site B plots by the end of the study, which may be a pollutant if the sediment is mobilised, although the spread of *S. anglica* on this site may help to prevent this. The pH was similar across all sites from the beginning of the study and was appropriate for saltmarsh plant development. SOC was highest on the SSSI as expected, but it did increase on other plots by 2017, after decreasing substantially the previous year due to the decline of the initial agricultural component along with sediment accretion and subsequent bare ground. The Site A plots had the most similar % SOC (and SOC tC/ha) to the SSSI by the end of the study. The results do indicate that on the sites which developed a characteristic salt marsh flora and an invertebrate fauna that began to resemble a salt marsh species composition, carbon storage rapidly approached that of a functional salt marsh, evidencing that the expected ecosystem service provisioning was developing. However, identification of the complete role in carbon sequestration needs further work. It is recommended that flux chambers and

mass spectrometry are used to assess whether the marsh is acting as a source or sink of CO_2 and to ascertain if denitrification is occurring. This can be assessed in parallel with carbon and nitrogen sampled in soil cores. New innovative methods are being used to assess how coastal wetlands are sequestering soil organic carbon, such as assessing how long it takes for tea bags to decompose in sediment (e.g. Mueller *et al.* 2018). Although the long-established use of litter bags has been widely used to assess carbon budgets, the tea bag method is cost-effective and allows for standardisation (See Keuskamp *et al.*, 2013). This method could be utilised on Steart Marsh to assess carbon sequestration rates in comparison to the reference sites. These data could also help to predict if the MR is likely to replace the role of the target state saltmarsh if this is eventually lost to sea level rise due to the effects of climate change and coastal squeeze.

Site B was characterised by long periods with standing water. Waterlogging can reduce sediment redox potential (Varty and Zedler, 2008; Sullivan *et al.*, 2018), and the standing water on the Site B plots during this study is thought to have partially influenced the lack of colonisation of plants on these plots. In future, redox potential should be considered alongside elevation on the MR sites on the Steart peninsula, because these two variables are associated (Davy *et al.*, 2011). The relationship of redox potential has been assessed in relation to elevation and plant species on British MR sites, and it was found that these environmental variables influence the colonisation of specific saltmarsh plant species (Sullivan *et al.*, 2018). It is therefore recognised that redox potential is an important variable to consider in future sampling on Steart Marsh and other MR sites, especially if water remains at lower elevations on study plots as it did in this study. It would also be of benefit to sample soil temperature because it can affect species interactions and is associated with salt levels in the soil (Bertness and Ewanchuk, 2002), although the effect of this may be limited in the temperate conditions of the UK. Additionally, testing soil cores for iron content would be beneficial because low concentrations of iron in the sediment can cause hydrogen sulphide to form which is toxic to plants (Weis and Butler, 2009). Testing these variables would add valuable information, but the analyses within this study already provides a useful indicator of how the marsh is developing in terms of plants, invertebrates, and soil characteristics.

To conclude, this research allows managers at Steart Marshes to understand how the marsh has developed ecologically with different starting states. These data can inform future management decisions on the site and stimulate further ideas for research. The information derived from analyses will also be valuable in the future design and management of

saltmarsh creation or restoration schemes, especially regarding compaction and starting states. It will be beneficial to understand these factors in more detail prior to creation or restoration work to enhance successful ecological functioning and facilitate rapid change toward desired target states.

7.1 Recommendations

From this research it is recommended that reducing levels of compaction prior to breaching on MR sites should be considered. Identifying areas of compaction from prior land use or engineering work during construction could be identified and methods such as subsoiling (depth of compaction will govern the type of machinery that will be required) could be incorporated to minimise levels of compaction. According to Spencer (2017) compaction in soil prior to breaching could be irreversible. If compaction is not reduced prior to breaching, draining areas of standing water in areas that are not initially designed to hold water is recommended to avoid anaerobic conditions that lead to a lack of typical halophytic plants and invertebrates colonising these areas. Standing water on the site could also lead to denitrification, although this needs further investigation on an MR site. It is also suggested that water is not held for long periods on Otterhampton Marsh if it is desirable for halophyte communities to develop. This may not be possible in certain areas due to the priority of managing bird populations such as waterfowl, but should be kept in mind if it is desirable to enable other ecosystem services associated with natural systems to develop, such as carbon sequestration. Additionally, if invertebrate abundance is low, such sites may not be managed in a way that maximises their potential to support wading birds dependent on soil invertebrates. Reducing submersion of soil may also minimise erosion of topsoil that occurred on this brackish site. Further ecological research on brackish schemes is recommended as there are limited studies on these sites compared to saltmarsh MR schemes.

It was determined that the heights of both plots on Site A were similar to one another after final elevation measurements were taken in 2018, and this was also apparent on Site B. Although all of the studied plots were within 0.5 metres AOD of each other at the start of the study, land heights of plots did differ from one another on each site prior to breaching. This suggests that the marsh is lacking topographical heterogeneity after accreting, which compares to findings by other researchers who have studied topography on MR sites (e.g. Brooks et al, 2015; Lawrence et al, 2018). It is therefore recommended that landscaping is considered in the future if there is a desire to create ecosystems that are comparable to natural marshes in terms of typical landscape heterogeneity.

Although typical saltmarsh plants are clearly colonising the site, it is recommended that planting experiments are conducted to assess how this influences invertebrate colonisation. It is clear from this research that invertebrates are colonising, and this is at least partially connected to the plant species that have colonised. Although many halophyte species have propagated in Steart, typical saltmarsh communities still may not develop due to the prevalence of bare ground in certain areas. Fully vegetated areas may facilitate an increase in invertebrate communities, especially if they contain similar plant composition to mature, natural saltmarshes. Planted plots could be compared to non-planted plots to assess if there are differences in invertebrate community composition.

As livestock grazing may have had an influence on plant community composition on Steart Marsh and Otterhampton Marsh, especially in relation to a mature marsh that is likely to be grazed by wildlife, such as geese, deer and lagomorphs, it would be of interest to compare ungrazed and grazed plots to assess the influence of cattle and sheep grazing on species communities. It is thought that cattle in particular can help to reduce *Spartina* biomass on marshes (Doody, 2008), so assessing the impact would be beneficial.

Continued assessment of the rapid colonisation of the invasive *P. marginata* will be of interest, especially because it can reduce biomass of *Spartina*. It has been used as a control method of cordgrass in America, and research suggests that it is spreading rapidly across coasts in the U.K. and may act as a control for *S. anglica* (Harkin, 2016). This may be beneficial for bird species on Steart Marsh due to the negative effect of *Spartina* on feeding grounds, but it is also important to note that this may affect the stability of marshes and lead to erosion. Monitoring of the species should be considered and *Spartina* biomass could be assessed alongside this to determine if it is having the same effect on the Steart MR scheme.

In conclusion, this study of the early stages of colonisation of a managed retreat clearly demonstrates that colonisation by plants and invertebrates are interlinked and that both are related to changes of soil conditions, which differs in relation to past land use, site engineering associated with managed retreat, and subsequent site management. Due to this, there are unanswered questions which warrant further research.

CHAPTER 8: REFERENCES

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